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Radian Corp., Research Triangle Park, NC

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This report is an analysis of potential human health risks and environmental effects associated with pollutants emitted to the air from both existing as well as planned municipal waste combustors in the U.S. Quantitative cancer risk assessment was restricted to direct inhalation of organic and metal emissions. Limited analysis of indirect exposure pathways, e.g., ingestion and dermal absorption, was performed to qualitatively determine the importance of this exposure relative to direct inhalation. Potential welfare effects were also evaluated. This effort was in support of a regulatory determination of municipal waste combustor emissions under the Clean Air Act. Pollutants considered in the analysis included: polychlorinated dioxins and dibenzofurans, chlorophenols, chlorobenzenes, formaldehyde, polychlorinated biphenyls, polycyclic aromatic hydrocarbons, arsenic, beryllium, cadmium, hexavalent chromium, lead, mercury, and hydrogen chloride.

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1.0 INTRODUCTION

This volume contains an analysis of potential health risk and environmental effects due to pollutants emitted from municipal waste combustion. The information presented herein was developed during a comprehensive, integrated study of municipal waste combustion. An overview of the findings of this study may be found in the Report to Congress on Municipal Waste Combustion (EPA/530-SW-87-021a). Other technical volumes issued as part of the municipal waste combustion study include:

- Emission Data Base for Municipal Waste Combustors (EPA/530-SW-87-021b)
- Combustion Control of Organic Emissions (EPA/530-SW-87-021c)
- Flue Gas Cleaning Technology (EPA/530-SW-87-021d)
- Costs of Flue Gas Cleaning Technologies (EPA/530-SW-87-021e)
- Sampling and Analysis of Municipal Waste Combustors (EPA/530-SW-87-021f)
- Characterization of the Municipal Waste Combustion Industry (EPA/530-SW-87-021h)
- Recycling of Solid Waste (EPA/530-SW-87-021i)

In view of the rapid growth predicted in the municipal waste combustion industry described in "Municipal Waste Combustion: Characterization of the Municipal Waste Combustion Industry," the U. S. Environmental Protection Agency (EPA) has a limited opportunity to prospectively evaluate the potential environmental and health impacts that may result from an expected proliferation of municipal waste combustion activity in the United States. In this regard, the agency is developing a methodology for the evaluation of emissions of pollutants into the atmosphere from the stacks of municipal waste combustors. The methodology consists of a series of environmental fate and transport models that utilize the known physical and chemical properties of specific pollutants to predict the atmospheric dispersion from stack emissions, the potential for surface deposition and accumulation; the movement of the settled pollutants through and into various environmental media; the potential bioaccumulation of pollutants into trophic systems; the potential for adverse effects on the vitality of natural ecosystems; and the potential for adverse effects on human health. The specific models used in this analysis are: the Human Exposure Model (HEM), the Industrial Source Complex Short-Term Air Dispersion Model, the Terrestrial Food Chain Model, the Surface Runoff Model, the Groundwater Infiltration Model, and the Dermal Exposure Model.

This report on risk analysis was preceded by a methodology document presented to the Science Advisory Board (SAB) for review in October 1986.[1] The document proposed a methodology for a multipollutant, multiple exposure pathway risk assessment and described mathematical models in detail. Review and comment by the SAB was favorable for the most part; however, the SAB recommended continued methodology development in some areas to enhance the indirect exposure modeling. In response to these comments, the EPA is continuing to develop and improve the models used to analyze indirect multiple exposure pathways. At the same time, the EPA is beginning to exercise the methodology to assess municipal waste combustor emissions, pollutant-by- pollutant.

The utility of the present methodology is limited by a number of gaps in the available technical data and significant uncertainties in many of the major analytical parameters, as well as by a need to perform further research to improve the methodology. For example, the evaluation of pollutants is limited by the relatively small number of organic and inorganic constituents that have been measured in municipal waste combustor emissions. Another constraint on the methodology is the limited amount of data regarding the physical and chemical behavior of specific pollutants in the natural environment, and the adverse impact these pollutants may have on human health.

The risk assessment described in this report relies on the present methodology to evaluate the effects of municipal waste combustor emissions from both existing facilities and from projected facilities that are anticipated to be built in the next ten to fifteen years. Direct and indirect exposure pathways were evaluated. The direct inhalation quantitative risk assessment considered potential human health risk based on carcinogenic and non- carcinogenic health effects from direct inhalation of the pollutants emitted from the stack. In addition, the direct exposure assessment evaluated other welfare impacts resulting from direct emissions to the atmosphere. The indirect exposure pathway considered adverse effects on human health and the environment resulting from exposure to emitted pollutants that ultimately deposit on soil. A significant limitation to those analyses is that only the impact of stack emissions was considered. Potential adverse health effects arising from the land disposal of solid residues from municipal waste combustors, e.g., fly ash and bottom ash were not addressed. However, the Agency is currently undertaking a regulatory analysis of ash disposal.

For the direct inhalation exposure assessment, actual plant data were used from existing facilities and a limited number of model plants were developed to represent projected new facilities. In addition, different combustor design types were taken into consideration in the analysis. For the indirect exposure assessment, two facilities were modeled using a reasonable worst-case analysis. One facility is an existing facility located in Virginia and the second is a projected model facility located in western Florida.

Thus far, preliminary indirect exposure analyses have been performed for only nine pollutants. Because of the preliminary nature of the exposure results and the continuing methodology development, the EPA believes it would be premature to produce quantitative risk results for indirect exposure pathways at this time. However, quantitative results for thirteen pollutants are discussed in terms of potential health and welfare effects due to direct emissions to the air.

The remainder of this report has been organized into two major sections and several technical appendices. Section 2.0 contains a summary of the results of the analyses of potential health risk and environmental effects, and Section 3.0 describes the methodology used in performing the analyses. Detailed descriptions of the models used in the analyses have been placed in appendices.

2.0 SUMMARY OF RESULTS

2.1 INTRODUCTION

This section summarizes the results of the preliminary risk analysis of pollutant emissions from municipal waste combustors. The analysis involved complex modeling of many different aspects of the industry and many of the pollutants emitted from those combustors. To enable the reader to better understand the results, a brief description of the modeling methodology and the parameters defining the system to be modeled has been included in Section 2.2 before the results are discussed. The approaches used to assess the potential effects resulting from direct and indirect exposure to municipal waste combustor emissions are described in detail. Section 2.3 discusses the results of the analysis of human health and welfare effects from direct exposure to pollutants from municipal waste combustors. Section 2.4 is a discussion of the general observations regarding the potential health and environmental effects from indirect exposure to deposited pollutants through groundwater, surface water, soil, and food. Finally, a discussion of the potential impact of emission control on reducing risk is presented in Section 2.5.

2.2 SUMMARY OF RISK ASSESSMENT METHODOLOGY AND PARAMETERS

The major parameters defining the systems modeled to estimate potential effects from municipal waste combustor emissions are shown in Table 2-1. It can be seen from the number of major parameters that the modeling results consist of thousands of numerical estimates.

Two pollutant pathways were modeled: the primary pathway, which is direct exposure to municipal waste combustor emissions emitted to the atmosphere; and a secondary pathway, which is indirect exposure to deposited pollutants. Pollutant concentrations were estimated in air (direct pathway) and in surface water, groundwater, soil, and food (indirect pathway through deposition).

TABLE 2-1. MAJOR PARAMETERS DEFINING THE EXPOSURE AND RISK ANALYSIS OF MUNICIPAL WASTE COMBUSTOR EMISSIONS

Exposure Modeling	Exposure Pathways	Populations of Combustors	Design Types	Control Scenarios	Pollutants
Direct Emission to the Atmosphere Indirect Exposure from Pollutant Deposition - surface water - groundwater - soil - food	Direct Inhalation Ingestion Dermal Absorption Plant Uptake	Existing Projected	Massburn Nonheat Recovery Massburn Heat Recovery Modular Refuse-derived Fuel (RDF)	Baseline Controlled	Organics chlorinated dibenzo-p-dioxins/chlorinated dibenzofurans (CDD/CDF) chlorophenols chlorobenzenes formaldehyde polychlorinated biphenyls (PCB) polycyclic aromatic hydrocarbons (PAH) Inorganics and Metals arsenic beryllium cadmium chromium (+6) nickel mercury lead particulate matter hydrogen chloride

For the direct inhalation pathway, human health and welfare effects were evaluated using the publicly reviewed and accepted Human Exposure Model (HEM). For the indirect pathway, potential effects on humans and the environment were evaluated using methodology currently being reviewed and developed: the Industrial Source Complex Air Dispersion Model, the Terrestrial Food Chain Model, the Surface Runoff Model, the Groundwater Infiltration Model and the Dermal Exposure Model.

2.2.1 Modeling for Direct Exposure Pathway

Modeling for the direct exposure pathway was performed for the existing population of combustors and for projected facilities, which are facilities anticipated to be constructed over the next 10 to 15 years. Furthermore, because the emissions from different types of combustors varied significantly, the analyses were performed for the various types of combustor design. Modeling the estimated pollutant concentrations in the atmosphere and possible effects from direct exposure to pollutants from the existing population of combustors was performed by using actual sizes, types, and locations of existing facilities. Air pollutant concentrations and possible direct exposure effects from projected municipal waste combustors were estimated using model plants located in representative urban, suburban, and rural locations.

Two control scenarios were modeled: 1) a baseline scenario, constructed for modeling purposes to represent the status quo (existing controls for the existing facility and electrostatic precipitators (ESPs) for the projected facilities) and 2) a controlled scenario representing uniform application of dry alkaline scrubbers and efficient particulate matter control (fabric filters) to all facilities.

The following types of human health and welfare effects were studied for the direct exposure pathway: cancer, noncancer health effects, and welfare effects. Noncancer health effects were evaluated by comparing modeled concentrations of pollutants with pollutant levels associated with adverse effects. Welfare effects were also estimated by comparison to threshold effects levels.

For the direct inhalation pathway, to calculate a probability of cancer, the exposure estimates obtained from the Human Exposure Model were combined with estimates of carcinogenic potency ("unit risk estimate"). The unit risk estimate for an air pollutant is defined by the EPA as a rough but plausible probability of the upper 95 percent confidence limit of the lifetime cancer risk that could occur in a population in which all individuals are exposed continuously from birth throughout their lifetimes (70 years) to a unit concentration (e.g., 1 ug/m^3) of the carcinogen in the air they breathe. Encompassed in the notion of upper bound is that the computed risk is likely to be higher than the true risk and that in fact, the true risk is probably much lower than the computed risk.

By combining the estimates of public exposure with the unit risk estimates, two measures of excess cancer risk were calculated: nationwide annual cancer incidence and maximum individual lifetime cancer risk. The annual cancer incidence is an estimate of cancer cases per year in the total population residing within 50 kilometers in all directions of the municipal waste combustor. The selection of 50 kilometers was an arbitrary modeling consideration. Maximum individual lifetime cancer risk is the probability that a person residing in the vicinity of the municipal waste combustor, and exposed continuously to the highest modeled annual average ambient air concentration of pollutant, will develop cancer over a 70-year lifetime.

2.2.2 Modeling for Indirect Exposure Pathway

The quantitative analysis of health impacts was limited to those from direct inhalation of municipal waste combustor emissions. The EPA has begun a preliminary analysis of the potential for exposure from the deposition of emitted pollutants and subsequent human contact through indirect exposure pathways, e.g., ingestion and dermal absorption. Because the Agency is continuing to develop the methodology and incorporate technical comments received during review by EPA's Science Advisory Board (SAB), the EPA believes it would be premature at this time to generate quantitative risk estimates from the indirect exposure modeling. The SAB believes it is necessary to validate the environmental fate parameters of specific chemicals used in the analysis before reliable estimates can be made of exposure and risk from ingestion of contaminated soil and foods and dermal contact with the soil. Furthermore, the SAB would like EPA to conduct studies to verify the computer simulation modeling of the movement of pollutants through the environment and into the human food chain. In view of these concerns, EPA decided only to employ the indirect exposure methodology to study whether prolonged stack emissions of pollutants that deposit over time onto the soil surface could significantly contribute to the total exposure to individuals living near a municipal waste combustor, and not to generate a risk assessment of this exposure.

Instead, of deriving quantitative risk estimates, the methodology was employed to test the hypothesis of whether indirect exposure pathways could contribute significantly to the total exposure to stack emissions from municipal waste combustors. To examine this possibility, two facilities were used to model possible effects. One was a hypothetical model plant arbitrarily located in western Florida devised to be representative of planned massburn energy recovery municipal waste combustors. The second facility, an existing facility located in Virginia, was employed to represent existing massburn energy recovery technology. Long-term deposition and exposures over 30 years and 100 years were modeled. Thirty years was chosen to represent the probable lifetime of a typical facility, and 100 years was selected with the assumption that once dedicated to that purpose, the same site would be used for successive replacement of municipal waste combustor units.

As discussed for the direct exposure pathway, two control scenarios were also modeled for the indirect pathways. The baseline scenario represented existing air pollution controls for the existing facility and an ESP for the hypothetical facility. The controlled scenario represented uniform application of dry alkaline scrubbers and efficient particulate matter control (fabric filter) to all facilities.

For the indirect pathway, the analysis focused on estimated pollutant concentrations and possible exposure levels. Because the methodology used to assess indirect exposure is currently under development, quantitative risk results for the indirect exposure pathway are not yet available and therefore have not been presented.

2.3 RESULTS OF DIRECT INHALATION EXPOSURE PATHWAY RISK ASSESSMENT

2.3.1 Human Carcinogenic Risks

Tables 2-2 and 2-3 summarize the estimated annual cancer incidence and maximum individual lifetime cancer risk (MIR) resulting from direct inhalation exposure to ambient air concentrations of pollutants emitted from the stack(s) of municipal waste combustors. In the tables, the annual incidence and MIR estimates are disaggregated by control scenario, existing and projected combustor populations, combustor technology, and pollutant class (organics or metals). Also shown are the annual cancer incidence and maximum individual risk (MIR) estimates contributed by pollutant classes and by category of municipal waste combustor design. For existing combustors, the massburn nonheat recovery category is associated with the highest risk, and for projected sources, the RDF units appear to pose the highest risk.

In Table 2-4, the ranges of the estimated cancer risk resulting from direct inhalation exposure to predicted ambient concentrations of modeled organic species are summarized. It is apparent from Table 2-4 that 31 to 98 percent of the total risk from direct inhalation exposure is attributable to stack emissions of CDD/CDF.

2.3.1.1 Ranges and Uncertainties. The risk ranges shown in Tables 2-2, 2-3, and 2-4 reflect several areas of uncertainty in the analysis. One area of uncertainty is the emissions data which were used to develop emission factors. Uncertainties associated with these data are discussed in detail in Section 2.2.1.4. For example, recovery of CDD/CDF from the Modified Method 5 stack sampling trains in some cases has been reported to be as low as 10 percent. Thus, actual emission levels could be higher than reported emissions by an order of magnitude. However, recovery has also been reported to be as high as 100 percent. The ranges shown incorporate recovery levels of 10 to 100 percent to account for variation in sampling method.

A basic inconsistency exists between emission factor data and toxicity data, both of which are needed to estimate risk. Emissions data are available for mixtures or classes of organic compounds. However, toxicity data have

TABLE 2-2. ESTIMATED NATIONWIDE CANCER RISK FROM DIRECT INHALATION EXPOSURE TO EMISSONS UNDER BASELINE CONTROL SCENARIO

Population of Municipal Waste Combustors	Organics[a]		Metals[b]		Combined	
	Ann. Incid.[c]	Max. Indiv.[d]	Ann. Incid.	Max. Indiv.	Ann. Incid.	Max. Indiv.
Existing Sources (1985)						
Massburn (Non-heat)	1.3 - 30	$10^{-4} - 10^{-3}$	.2	10^{-5}	1.5 - 30	$10^{-4} - 10^{-3}$
Massburn (Heat Rec)	.2 - 4	$10^{-4} - 10^{-3}$	.04	10^{-4}	.2 - 4	$10^{-4} - 10^{-3}$
RDF	.1 - 3	$10^{-5} - 10^{-3}$	.2	10^{-5}	.3 - 3.2	$10^{-5} - 10^{-3}$
Modular	.0008-.01	$10^{-6} - 10^{-4}$	.01	10^{-4}	.01 - .02	$10^{-4} - 10^{-4}$
TOTAL ESTIMATED RISK FROM EXISTING SOURCES (ROUNDED)[e]	2 - 40	$10^{-4} - 10^{-3}$	.5	10^{-4}	2 - 40	$10^{-4} - 10^{-3}$
Projected Sources (1993)						
Massburn (Heat Rec)	.3 - 7	$10^{-6} - 10^{-5}$	.3	10^{-6}	.6 - 7.3	$10^{-6} - 10^{-5}$
RDF	.8 - 14.2	$10^{-5} - 10^{-4}$	.1	10^{-6}	.9 - 14.3	$10^{-5} - 10^{-4}$
Modular	.04 - .9	$10^{-6} - 10^{-5}$	.01	10^{-6}	.05 - .9	$10^{-6} - 10^{-5}$
TOTAL ESTIMATED RISK FROM PROJECTED SOURCES (ROUNDED)[e]	1 - 20	$10^{-5} - 10^{-4}$	.4	10^{-6}	2 - 20	$10^{-5} - 10^{-4}$
COMBINED TOTAL (ROUNDED)[e]	3 - 60	$10^{-4} - 10^{-3}$	.9	10^{-4}	4 - 60	$10^{-4} - 10^{-3}$

[a] CDD/CDF, chlorophenols, chlorobenzenes, formaldehyde, PCB, PAH. Organic emissions are based on assumed 20% control efficiency for both existing and projected source air pollution control equipment.

[b] Arsenic, beryllium, cadmium, chromium +6. Metal emissions are based on estimated efficiency for installed air pollution control equipment for existing facilities, and a 99% efficient ESP for projected facilities.

[c] Annual incidence is the aggregate risk of cancer cases per year in populations within 50 km of all municipal waste combustors in the U.S.

[d] Maximum individual risk is the probability that a person exposed to the highest modeled concentration of pollutants from a municipal waste combustor to which anyone is exposed will develop cancer over a 70-year lifespan.

[e] Apparent errors in total are due to intentional rounding to one significant figure.

TABLE 2-3. IMPACT OF THE APPLICATION OF DRY SCRUBBER/FABRIC FILTER CONTROL DEVICES ON THE ESTIMATED NATIONWIDE CANCER RISK FROM DIRECT INHALATION EXPOSURE TO EMISSIONS

Population of Municipal Waste Combustors	Organics[a]		Metals[b]		Combined	
	Ann. Incid.[c]	Max. Indiv.[d]	Ann. Incid.	Max. Indiv.	Ann. Incid.	Max. Indiv.
Existing Sources (1985)						
Massburn (Non-heat)	.08 - 1.9	10^{-5} - 10^{-4}	.05	10^{-6}	.1 - 2	10^{-5} - 10^{-4}
Massburn (Heat Rec)	.01 - .3	10^{-5} - 10^{-4}	.01	10^{-6}	.02 - .3	10^{-5} - 10^{-4}
RDF	.01 - .2	10^{-6} - 10^{-5}	.03	10^{-6}	.04 - .2	10^{-6} - 10^{-5}
Modular	<.00001	10^{-7} - 10^{-6}	.001	10^{-6}	.001	10^{-6} - 10^{-6}
TOTAL ESTIMATED RISK FROM EXISTING SOURCES (ROUNDED)[e]	.1 - 2.4	10^{-5} - 10^{-4}	.1	10^{-6}	.2 - 2.5	10^{-5} - 10^{-4}
Projected Sources (1993)						
Massburn (Heat Rec)	.02-.4	10^{-7} - 10^{-6}	.2	10^{-6}	.2 - .6	10^{-6} - 10^{-6}
RDF	.05 - 0.9	10^{-6} - 10^{-5}	.04	10^{-6}	0.09-0.9	10^{-6} - 10^{-5}
Modular	.002-.04	10^{-7} - 10^{-6}	.001	10^{-6}	.003-.03	10^{-7} - 10^{-6}
TOTAL ESTIMATED RISK FROM PROJECTED SOURCES (ROUNDED)[e]	0.07-1.3	10^{-6} - 10^{-5}	.2	10^{-6}	0.3-1.5	10^{-6} - 10^{-5}
COMBINED TOTAL (ROUNDED)[e]	.2 - 4	10^{-5} - 10^{-4}	.3	10^{-6}	.5 - 4.0	10^{-5} - 10^{-4}

[a] CDDs/CDFs, chlorophenols, chlorobenzenes, formaldehyde, PCB, PAH. Organic emissions are based on 99.5 percent control efficiency for dry scrubber/fabric filter controls.

[b] Arsenic, beryllium, cadmium, chromium +6. Metal emissions are based on 95% control efficiency for dry scrubber/fabric filter controls.

[c] Annual incidence is the aggregate risk of cancer cases per year in population within 50 km of all municipal waste combustors in the U.S.

[d] Maximum individual risk is the probability that a person exposed to the highest modeled concentration of pollutants from a municipal waste combustor to which anyone is exposed will develop cancer over a 70-year lifespan.

[e] Apparent errors in total are due to intentional rounding to one significant figure.

TABLE 2-4. RANGES IN THE CONTRIBUTION OF POLLUTANTS IN MUNICIPAL WASTE COMBUSTOR (MWC) EMISSIONS TO THE ESTIMATED TOTAL ANNUAL CANCER INCIDENCE AND MAXIMUM INDIVIDUAL LIFETIME CANCER RISK

Pollutant	Existing MWC Annual Cancer Incidence[a,b]	Existing MWC Maximum Individual Risk Range[c,d]	Projected MWC Annual Cancer Incidence[a,b]	Projected MWC Maximum Individual Risk Range[c,d]
Chlorinated dioxins and dibenzofurans	2 to 40	10^{-6} to 10^{-3}	0.8 to 20	10^{-6} to 10^{-4}
Chlorophenols	0.0001 to 0.0003	10^{-9} to 10^{-8}	0.0001 to 0.0003	10^{-10} to 10^{-9}
Chlorobenzenes	0.009 to 0.02	10^{-7} to 10^{-6}	0.004 to 0.01	10^{-9} to 10^{-7}
Formaldehyde	0.009	10^{-8}	0.02	10^{-8} to 10^{-7}
Polycyclic aromatic hydrocarbons	0.01 to 0.6	10^{-7} to 10^{-5}	0.05 to 3.0	10^{-7} to 10^{-5}
Polychlorinated biphenyls	0.02	10^{-8} to 10^{-5}	0.2	10^{-9} to 10^{-6}
Arsenic	0.2	10^{-7} to 10^{-4}	0.1	10^{-8} to 10^{-7}
Beryllium	0.02	10^{-9} to 10^{-6}	0.001	10^{-11} to 10^{-8}
Cadmium	0.2	10^{-6} to 10^{-4}	0.2	10^{-7} to 10^{-6}
Chromium	0.2	10^{-7} to 10^{-4}	0.1	10^{-8} to 10^{-6}
Rounded Total[e]:	2 to 40	10^{-6} to 10^{-3}	2 to 20	10^{-6} to 10^{-4}

[a] The ranges in annual cancer incidence reflect the assumptions made regarding the potential carcinogenicity of classes of organic compounds.

[b] Annual cancer incidence is defined as the average number of excess cancer cases experienced annually in exposed populations.

[c] The ranges in maximum individual lifetime cancer risk reflect differences in emissions and the evaluation of emissions from municipal waste combustor technologies within the existing and projected categories.

[d] Maximum individual risk is defined as the probability of contracting cancer following lifetime exposure at the maximum modeled long-term ambient concentration. Probability is expressed as a negative exponent of 10. A risk of one chance in 10,000 is expressed as 10^{-4}.

[e] Apparent errors in total are due to intentional rounding to one significant figure.

been developed with respect to human exposure to discrete organic compounds. Therefore, toxicity data are not available for all the compounds within a class. Nevertheless, scientists generally agree that structurally related compounds may exhibit similar toxic effects through similar mechanisms of toxicity. Therefore, some assumptions are necessary to relate the emissions data to the risk measures and to account for the potential toxicity of various compounds in a mixture to which the general population may be exposed. In the case of CDD/CDF, a method involving toxic equivalency factors (TEF) was used to convert emissions of a mixture of CDD and CDF congeners to an equivalent quantity of the most toxic compound in the class, 2,3,7,8-TCDD. The method for conversion, using weighting factors based on relative toxicities, has been adopted by the EPA as an interim procedure to evaluate mixtures of CDD/CDF (52 FR 11749).[1] Table 2-5 shows TEFs recommended in EPA's interim procedures.

In applying the TEF method to CDD/CDF emissions, another discrepancy between emissions data and toxicity data had to be resolved. TEFs were devised with the assumption that certain CDD/CDF isomers having chlorines in the 2,3,7,8 positions on the benzene rings are the most toxic. However, most emissions data are in terms of homologs. Emission tests on a few facilities have reported data on the emission levels of specific isomers. When TEF calculations based on isomer-specific data are compared to calculations based on homolog data at the same facility, the calculations based on homolog data are usually higher by a factor of 3 to 7. Although it is unclear whether this same pattern would hold for those facilities for which only homolog-specific data are available, the ranges shown for CDD/CDF have incorporated these factors of 3 to 7.

Additional inconsistencies between emissions data and toxicity data exist for other organics as well. For the classes of polycyclic aromatic hydrocarbons (PAH), chlorobenzenes, and chlorophenols, cancer potency estimates have been established only for specific compounds within each class: benzo(a)pyrene (a specific PAH), hexachlorobenzene, 2,2,4-trichlorophenol and aroclor 1254 PCB. However, emission data are available only for the class of compounds as a whole (e.g., chlorobenzenes). Therefore, ranges were formed based on assumptions of the relative carcinogenicity of each class compared to the carcinogenicity of the individual compounds. Using chlorobenzenes as an

TABLE 2-5. EPA-RECOMMENDED TOXIC EQUIVALENCY FACTORS FOR MIXTURES OF CHLORINATED DIBENZO-P-DIOXINS (CDDs) AND -DIBENZOFURANS (CDFs)

	Toxic Equivalency Factors
CDD	
Mono thru di	0
Tri-CDD	0
2,3,7,8-TCDD	1
Other TCDDs	0.01
2,3,7,8-PentaCDD	0.5
Other PentaCDDs	0.005
2,3,7,8-HexaCDD	0.04
Other HexaCDDs	0.0004
2,3,7,8-HeptaCDD	0.001
Other HeptaCDDs	0.00001
OctaCDD	0
CDF	
2,3,7,8-TCDF	0.1
Other TCDFs	0.001
2,3,7,8-PentaCDF	0.1
Other PentaCDFs	0.001
2,3,7,8-HexaCDF	0.01
Other HexaCDFs	0.0001
2,3,7,8-HeptaCDF	0.001
Other HeptaCDFs	0.00001
OctaCDF	[illegible]

example, the high risk estimate represents the assumption that all chlorobenzenes have cancer potencies equal to that of hexachlorobenzene, which, based on test data, is assumed to be 43 percent of the mixture. The low estimate is based on the assumption that only the single compound, hexachlorobenzene (of the chlorobenzenes) is associated with any risk of cancer.

The range of risk estimates from direct inhalation of formaldehyde reflects the uncertainty surrounding which tumors induced in laboratory animals by exposure to formaldehyde are indicative of formaldehyde's potential for causing cancer in humans. The low estimate is based on the assumption that only some of the tumors in animals are indicative of formaldehyde's potential for causing cancer in humans, and the high estimate is based on the assumption that all of the tumors are indicative of formaldehyde's potential for causing cancer in humans.[2]

Still another area of uncertainty has not been incorporated in the risk ranges shown. A significant portion (80 percent or more) of the organic emissions from the stacks of municipal waste combustors have not been identified and quantified. Although some portion of the mixture may be carcinogenic, the carcinogenic fraction, its composition, and its potency remain unknown. If the unspeciated organics had a carcinogenic potency equivalent to the average potency of those compounds evaluated, even excluding CDD/CDF, the contribution to the annual incidence estimates could be appreciable. However, there is no information to quantify this potential source of risk.

2.3.2 Direct Exposure Noncarcinogenic Risks of Lead and Mercury Emissions

The potential for noncarcinogenic health risk of lead and mercury emitted to the atmosphere and subsequently inhaled was evaluated by comparing maximum modeled estimated annual average concentrations to standards or guidelines. For lead, the modeled concentrations were compared to the ambient air quality standard (NAAQS) of 1.5 microgram per cubic meter of air (ug/m^3) (40 CFR 50.12). Modeled mercury concentrations were compared to the National Emission Standard for Hazardous Air Pollutants (NESHAP) guideline of 1 ug/m^3. Results of the comparisons are shown in Tables 2-6 and 2-7. The concentrations presented in the table are the maximum concentrations modeled for each class of combustor.

As shown in Table 2-6, the modeling results predicted ambient lead concentrations resulting from municipal waste combustors emissions to be less than the ambient lead standard. The modular category had the highest maximum modeled concentration of 0.9 ug/m^3, which is 60 percent of the ambient air quality standard. Individual municipal waste combustors are evaluated during the permitting process to ensure conformance to the lead standard, including consideration of background lead concentrations.

Table 2-7 displays the maximum annual average ground level concentrations of mercury modeled near municipal waste combustors. Estimated ambient air concentrations of mercury were based on total particulate mass emissions. Using controlled particulate emission estimates (measured downstream of an ESP) to predict mercury concentrations yielded maximum modeled ambient air concentrations for mercury that do not exceed the NESHAP guideline. The massburn design using a waterwall-type boiler for heat recovery had the highest maximum modeled concentration of 0.4 ug/m^3, which is 40 percent of the NESHAP guideline.

2.3.3 Human Welfare Effects

Among the pollutants found in stack gases from municipal waste combustors are acid gases. The major acid specie of concern to the EPA is hydrochloric acid (HCl) because of the magnitude of emissions. Short-term and long-term modeled concentrations of HCl in the vicinity of existing and projected sources are shown in Table 2-8.

The estimated ambient concentrations were compared to an annual average concentration of 3.0 ug/m^3 which is associated with materials damage in the form of corrosion of ferrous metals.[3] The comparisons showed that under the baseline control scenario, maximum annual average HCl concentrations predicted for over 50 percent of the existing municipal waste combustors, exceeded the materials damage level. The modeling results for the projected facilities under the baseline scenario indicated that long-term concentrations may exceed the 3.0 ug/m^3 level for the larger capacity massburn and RDF units and for all capacities of modular units, depending on location and local meteorological conditions.

TABLE 2-6. COMPARISON OF MAXIMUM MODELED CONCENTRATIONS OF LEAD TO AMBIENT STANDARD FOR EXISTING MUNICIPAL WASTE COMBUSTORS

Design Type	Modeled Maximum Annual Average Lead Concentration[a] (ug/m^3)[c]	Maximum Modeled Conc./ Ambient Standard[b] (%)
Massburn Nonheat Recovery	0.6	40
Massburn Heat Recovery	0.8	53
Massburn Heat Recovery-Refractory	0.3	20
RDF	0.4	27
Modular - Nonheat Recovery	0.4	27
Modular - Heat Recovery	0.9	60

[a] Based on controlled particulate emissions modeled for existing combustors.

[b] National ambient air quality standard for lead is 1.5 ug lead/m^3 (quarterly average basis).

[c] ug/m^3 = micrograms of pollutant per cubic meter of air.

TABLE 2-7. COMPARISON OF MAXIMUM MODELED CONCENTRATIONS OF MERCURY TO NESHAP GUIDELINE FOR EXISTING MUNICIPAL WASTE COMBUSTORS

Design Type	Modeled Maximum Annual Average Mercury Concentration[a] (ug/m^3)[c]	Maximum Modeled Concentration/ NESHAP Guideline[b] (%)
Massburn Nonheat Recovery	0.1	10
Massburn Heat Recovery-Waterwall	0.4	40
Massburn Heat Recovery-Refractory	0.03	3
RDF	0.03	3
Modular - Nonheat Recovery	0.01	1
Modular - Heat Recovery	0.04	4

[a] Based on percentage of mercury in controlled particulate emission estimates (measured downstream from an ESP) for existing combustors.

[b] NESHAP guideline for mercury is 1 ug mercury/m^3 (30-day average).

[c] ug/m^3 = micrograms of pollutant per cubic meter of air.

TABLE 2-8. ESTIMATED AMBIENT AIR HCl CONCENTRATIONS MODELED IN THE VICINITY OF MUNICIPAL WASTE COMBUSTORS

Sources	Control Level	Total Annual HCl Emissions (Mg/yr)	Range of Predicted Annual Average Maximum Conc.[a] (ug/m^3)[b]	Range of Predicted Maximum 1-Hour Concentration[a] (ug/m^3)[b]
EXISTING SOURCES	Baseline	44,900	.1 - 68	64 - 2,500
	Controlled	4,490	.01 - 7	6 - 250
PROJECTED SOURCES	Baseline	194,400	.7 - 88	110 - 170
	Controlled	19,400	.07 - 9	11 - 17

[a] Ranges of HCl concentrations reported for all modeled municipal waste combustion facilities.

[b] ug/m^3 = micrograms of pollutant per cubic meter of air.

2.4 RESULTS OF INDIRECT EXPOSURE PATHWAY ANALYSIS

2.4.1 Indirect Exposure to Muncipal Waste Combustor Emissions

The indirect exposure analysis was intended to evaluate whether indirect exposure pathways could contribute significantly to the total exposure to municipal waste combustion emissions. The indirect exposure analysis complements the traditional direct exposure analysis by adding the exposure pathways of ingestion and dermal contact of deposited air emissions, as well as estimates of possible accumulation of contaminants into the natural environment. Such a comparison of indirect exposure potential to direct exposure levels has been done previously in analyses of exposure to 2,3,7,8-TCDD.[4]

The methodology for modeling indirect exposure to pollutants emitted to the atmosphere has received favorable review by the EPA's Science Advisory Board. At this time, however, the Board's technical comments on the methodology have not been fully incorporated into the various models. Also, chemical fate parameters selected for use in the indirect exposure models were from the published literature, but they have not been peer reviewed for this use. Given the preliminary nature of the methodology and assumptions, the EPA cannot interpret the results of the indirect exposure analysis quantitatively at this time. However, results of the indirect exposure analysis are summarized qualitatively in Tables 2-9, 2-10, and 2-11.

The objective of the analysis was to determine whether indirect exposures could contribute significantly to the total exposure due to municipal waste combustor emissions, therefore, the analysis used conservative, potentially worst-case, assumptions. For example, the analysis defined the most exposed individuals as a hypothetical farm family that obtained much of their food supply from the area of maximum deposition of pollutants emitted from the stack of the two modeled municipal waste combustors, and whose children ingested 0.5 grams of soil per day. This family was assumed to live just outside the estimated boundary (200 meters, or about one-tenth of a mile) of the two modeled combustion facilities. The preliminary conclusions concerning potential exposures must be interpreted in light of this worst-case, but still

TABLE 2-9. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL HUMAN HEALTH EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS
(Exposure by Ingestion - Inorganic Compounds)

Facility	Control Scenario	Deposition Interval	Food Ingestion			Soil Ingestion			Surface Water Ingestion			Fish Ingestion		
			Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.		Pb, Cr Hg, Ni	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
(VIRGINIA-120 tpd)		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	PB, Ni Cr, Be	
	Controlled	30 yr.		Pb, Cr Hg, Ni	Be		Pb, Cr Hg, Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
Projected	Baseline	30 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
(WESTERN FLORIDA-3,000 tpd)		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
	Controlled	30 yr.	Hg	Pb, Ni Cr	Be	Hg	Pb, Ni Cr, Be			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	

Be = Beryllium   Cr = Chromium   Hg = Mercury   Ni = Nickel   Pb = Lead

TABLE 2-10. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL HUMAN HEALTH EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS
(Human Risk by Ingestion - Organic Compounds)

Facility	Control Scenario	Deposition Interval	Food Ingestion			Soil Ingestion			Surface Water Ingestion			Fish Ingestion		
			Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
(VIRGINIA-120 tpd)		100 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
	Controlled	30 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
		100 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
Projected	Baseline	30 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD CB PCB	FA	
(WESTERN FLORIDA-3,000 tpd)		100 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD CB PCB	FA	
	Controlled	30 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA	
		100 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA	

CB = Chlorobenzenes FA = Formaldehyde PCB = Polychlorinated benzenes PCDD = Polychlorinated dibenzo-p-dioxins

TABLE 2-11. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL ECOLOGICAL EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS (Inorganic Compounds)

			Ecological Risks					
			Terrestrial Impacts			Aquatic Impacts		
Facility	Control Scenario	Deposition Interval	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.		Pb, Cr Hg, Ni	Cr* Be	Pb Hg	Cr, Be Ni	Cr*
(VIRGINIA-120 tpd)		100 yr.	Pb Hg	Cr Ni	Cr Be	Pb Hg	Cr, Be Ni	Cr
	Controlled	30 yr.		Pb, Ni Cr	Cr Be	Pb Hg	Cr, Be Ni	Cr
		100 yr.	Hg	Pb, Ni Cr	Cr Be	Pb Hg	Cr, Be Ni	Cr
Projected	Baseline	30 yr.	Pb Hg	Pb, Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr
(WESTERN FLORIDA-3000 tpd)		100 yr.	Pb Hg	Cr, Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr
	Controlled	30 yr.	Hg	Pb, Ni Cr	Cr Be	Hg	Pb, Ni Cr, Be	Cr
		100 yr.	Pb	Cr Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr

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* Data on chromium were available for certain terrestrial and aquatic pathways showing low likelihood of effects, but were not available for other pathways (e.g. toxicity thresholds for soil biota and for predators of aquatic organisms).

Be = Beryllium Cr = Chromium Hg = Mercury Ni = Nickel Pb = Lead

plausible exposure scenario. The general population exposure to deposited emissions is likely to be less, possibly significantly less, in most situations.

Subject to the above assumptions and uncertainties, the preliminary indirect exposure analysis indicates that indirect exposure to long-term deposited emissions may be comparable to direct inhalation exposure for some constituents such as CDD/CDF, PCB, and chlorobenzenes.

Preliminary analysis indicates that for mercury emissions the principal indirect exposure pathway is through food ingestion, for example, ingestion of mercury-contaminated freshwater fish. Also, children in this hypothetical maximum exposure scenario who are assumed to ingest 0.5 grams of soil per day may be exposed to levels of deposited lead and mercury emissions that may indicate a potential health effect.

The preliminary analysis also indicated that indirect exposures to some pollutants were not of significant concern. The analysis found that indirect exposure to nickel, chromium (+6), beryllium, and formaldehyde would not significantly add to the inhalation exposure under the scenarios and time frames modeled.

2.4.2 Potential Environmental Effects

The indirect exposure modeling results indicated that both lead and mercury may reach levels in the environment of concern to the vitality of aquatic and terrestrial plant and animal life. In Sweden, the atmospheric deposition of mercury emissions has caused mercury levels in lakes in southern and central Sweden to increase by a factor of five, and mercury levels in northern lakes to have doubled.[5] As a result, Swedish authorities have restricted the consumption of some fresh water fish. Swedish authorities have attributed this increase of mercury in fresh water systems to emissions of mercury from stationary combustion sources, including municipal waste combustors, which essentially corroborates EPA's concerns.

2.5 POTENTIAL REDUCTION OF CARCINOGENIC RISK, WELFARE EFFECTS, AND INDIRECT EXPOSURE WITH IMPROVED EMISSION CONTROL

Predicted carcinogenic health risks due to direct emissions to the atmosphere under two control scenarios are compared in Tables 2-2 and 2-3. As indicated by the tables, maximum individual lifetime risk and annual cancer incidence is predicted to be reduced by about an order of magnitude with the uniform application of dry alkaline scrubbers combined with fabric filters when compared to the baseline control estimates. Table 2-12 shows the potential reductions in HCl concentration under the same baseline and control scenarios. Reductions of approximately 90 percent can be seen in predicted HCl concentrations with the use of dry alkaline scrubbers. This reduction in HCl emissions would bring ambient air concentrations near most municipal waste combustor facilities below levels associated with the corrosion of ferrous metals.

Indirect exposure to deposited pollutants would also be expected to be reduced by over an order of magnitude. However, as Table 2-9 (previously presented) shows, even under the controlled scenario, indirect exposures to lead and mercury cannot be dismissed for some combinations of soil, groundwater and surface water characteristics. Because there remains significant uncertainty concerning the indirect exposure results, exposures due to these pathways are being investigated further.

It is feasible to reduce mercury emissions from municipal waste combustors by 80 to 90 percent with the application of dry alkaline scrubbers in combination with fabric filters or electrostatic precipitators if the devices are operated at temperatures below 145^{o}C. The controlled scenario modeled for this analysis, however, assumed only 50 percent control of mercury.

TABLE 2-12. POSSIBLE REDUCTIONS IN AMBIENT AIR HCl CONCENTRATIONS WITH APPLICATION OF DRY ALKALINE SCRUBBERS

	Existing Municipal Waste Combustors	Projected Municipal Waste Combustors
Baseline Estimated Maximum Long-Term HCL Concentration ($\mu g/m^3$)	0.1 - 68	0.7 - 88
Estimated Maximum Long-Term HCl Concentration Under Controlled Scenario ($\mu g/m^3$)	0.01 - 7	0.1 - 9
Percent Reduction Achievable (%)	90	90

2.6 REFERENCES FOR SECTION 2.0

1. U. S. Environmental Protection Agency. Interim Procedures for Estimating Risks Associated with Exposure to Mixtures of Chlorinated-Dibenzo-p-dioxins and -Dibenzofurans (CDD and CDF). The Risk Assessment Forum, Washington, D.C. EPA 625/3-87-012, March 1987.

2. U. S. Environmental Protection Agency. Assessment of Health Risks to Garment Workers and Certain Home Residents from Exposure to Formaldehyde. Washington, D.C. March 1987.

3. Letter from Henry, J. (Texas Air Control Board), to Kellam, R. (U.S. Environmental Protection Agency). June 26, 1987. Information on ambient levels of HCl associated with corrosion.

4. U. S. Environmental Protection Agency. Estimating Exposures to 2,3,7,8-TCDD. Draft Report. June 1, 1987.

5. Energy from Waste. A study concerning technology, environment and economics conducted by the National Energy Authority and the National Environmental Protection Board for the Swedish Government. June 11, 1986.

3.0 METHODOLOGY

3.1 INTRODUCTION AND SUMMARY OF METHODOLOGY

This section explains the methodology used in the risk assessment of emissions from municipal waste combustors. In this analysis, emissions test data from three major types of municipal waste combustors were used to develop emission factors representative of emissions from typical municipal waste combustors. The emission factors were used to estimate emissions from both existing and projected facilities. Exposure assessments were then performed using a variety of mathematical models.

Exposure to emitted pollutants is influenced by many factors. Pollutants emitted from municipal waste combustors into the atmosphere may be distributed across environmental media (the atmosphere, soil and water) as a result of complex mechanisms, most of which are just beginning to be understood. Some pollutants are emitted in the combustion gas adsorbed onto the surface of particulate matter, while other pollutants remain in a gaseous state. Some pollutants may photodegrade, undergo complex chemical reactions, or combine with other pollutants when transported through the atmosphere. Particulate-bound pollutants may ultimately settle on the earth's surface by the forces of gravity. Precipitation passing through the plume may increase the rate of surface deposition by washout of adsorbed and gaseous pollutants. Once deposited on the surface the pollutants may again be physically, chemically, or photolytically transformed, may persist, or may be transported by the action of wind and precipitation to other environmental compartments. Deposited materials may even be resuspended into the atmosphere, and once again become air pollutants. The net effect is that human exposure to municipal waste combustor emissions results not only from inhalation of direct emissions, but also indirectly from ingestion of deposited pollutants in soil, water and food, and from skin contact with deposited pollutants in soil.

Detailed experimental evaluation of the environmental fate and environmental transport of municipal waste combustor emissions have not been done under actual conditions. Therefore, mathematical models of the fate and transport of pollutants entrained in the stack exhaust gas are currently the most feasible alternative to the assessment of human exposure to municipal waste combustor emissions. These models can also be used to estimate bioaccumulation in the natural ecosystem, potential accumulation of pollutants adverse to the promotion of animal and plant life, and accumulation of pollutants into the human food chain. The models specifically used in this analysis include: the Human Exposure Model (HEM), the Industrial Source Complex (ISC) Dispersion Model, the Terrestrial Food Chain Model, the Surface Runoff Model, the Groundwater Infiltration Model, and the Dermal Exposure Model.

Ambient concentrations of pollutants in the atmosphere surrounding municipal waste combustors and the associated levels of human exposure were modeled using the Human Exposure Model (HEM). The Industrial Source Complex (ISC) Model was used to estimate deposition of pollutants on surfaces surrounding municipal waste combustors. The potential exposures to humans from ingestion of soil and food contaminated by deposited municipal waste combustor emissions was estimated using the Terrestrial Food Chain Model. The Terrestrial Food Chain Model was also used to predict adverse effects on plants, herbivorous animals, and soil organisms from deposited pollutants. The potential human exposure to contaminated surface or groundwaters and from ingestion of fish from these contaminated waters was estimated using the Surface Runoff and Groundwater Infiltration Models. These models were also used to predict adverse effects on aquatic organisms living in contaminated waters. Finally, the potential exposures to humans from dermal contact with contaminated soils were estimated using the Dermal Exposure Model.

The probability of cancer from direct inhalation was estimated by applying the carcinogenic potencies for individual pollutants emitted from municipal waste combustors to the corresponding population exposure levels predicted by the HEM. Next, noncancer effects resulting from exposure to direct municipal waste combustor emissions to the air were also evaluated by comparing ambient concentrations of pollutants to levels associated with adverse health effects. Finally, the hypothesis that indirect exposure routes could contribute significantly to the total exposure to municipal waste combustion emissions was tested.

In addition to human exposure, the potential for adverse ecological and welfare effects associated with municipal waste combustor emissions was examined. These included material corrosion due to acid gas emissions, and levels of pollutants to which herbivorous animals, plants, soil organisms and aquatic organisms may be exposed.

The remainder of this section describes the methodology used to 1) estimate municipal waste combustor emissions, and 2) estimate exposure to, and evaluate the potential effects from, exposure to these emissions.

Section 3.2 presents the methodology used to estimate combustor emissions based on emission factors for the various pollutants. Section 3.3 describes the methodology used to estimate direct and indirect exposure to municipal waste combustors and to evaluate the effects of this exposure on human health and welfare and the environment. Direct inhalation exposure and indirect exposure and the associated modeling efforts are discussed in Sections 3.3.1 and 3.3.2, respectively. The methodology for evaluating ecological exposures from long-term deposition of pollutants is also presented in Section 3.3.2.

3.2 ESTIMATING EMISSIONS

While many compliance tests have been conducted for particulate emissions, a limited number of emission tests for specific pollutants have been conducted on municipal waste combustors in the United States, Canada, Western Europe, and Japan. These tests provide the available data on emissions of pollutants from the three major types of municipal waste combustors (i.e., massburn, refuse-derived fuel (RDF), and modular). Table 3-1 summarizes the number of facilities from which emissions data for various pollutants were available for use in this analysis.* Table 3-2 lists thirteen of the pollutants in municipal waste combustor emissions that were evaluated in the risk assessment.

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*Because the regulatory analysis required a timely assessment, an early version of the Emissions Data Base was used for the risk analysis. The earlier draft was publicly released in January, 1987.

TABLE 3-1. SUMMARY MATRIX OF EMISSIONS TEST DATA FOR MUNICIPAL WASTE COMBUSTORS USED IN THIS ANALYSIS

Pollutant	Number of Facilities Tested			
	Massburn	RDF	Modular	Total
Uncontrolled PM	10	3	3	16
Controlled PM	13	1	1	15
Metals				
Arsenic	6	2	3	11
Beryllium	5	1	1	7
Cadmium	9	1	2	12
Chromium	9	2	2	13
Mercury	5	2	2	9
Nickel	7	2	2	11
Lead	10	2	2	14
Acid Gases				
HCl	10	5	2	17
HF	5	1	2	8
Organics				
Chlorinated Dibenzo-p-dioxins/Chlorinated Dibenzofurans	8	5	6	19
Polycyclic Aromatic Hydrocarbons (PAH)	1	1	1	3
Polychlorinated Biphenyls (PCB)	2	2	2	6
Formaldehyde	2	3	2	7
Chlorobenzenes	2	2	1	5
Chlorophenols	2	2	1	5

Source: Reference 1

TABLE 3-2. POLLUTANTS EVALUATED IN THE RISK ASSESSMENT

- Arsenic
- Beryllium
- Cadmium
- Chlorobenzenes
- Chlorophenols
- Chromium^{+6}
- Chlorinated Dibenzo-p-dioxins and Chlorinated Dibenzofurans (CDD/CDF)[a]
- Formaldehyde
- Hydrogen chloride
- Lead
- Mercury
- Polychlorinated biphenyls (PCB)
- Polycyclic aromatic hydrocarbons (PAH)

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[a] The terms dioxins and dibenzofurans refer to a group of 75 chlorodibenzo-p-dioxin compounds and 135 chlorodibenzofuran compounds, each having similar chemical and physical properties.

For the most part, emissions of particulate matter (PM) from municipal waste combustors have been well characterized. Measurements of PM emissions have been made at a number of facilities representing a wide range of combustion technologies. Metals and acid gas emissions from conventional massburn combustors are also relatively well characterized; however, less data are available for metals and acid gas emissions from other combustor types. The most significant data gap in the inventory of municipal waste combustor emissions is the speciation of halogenated and nonhalogenated organic compounds. For some of the organic pollutants in particular, data are limited to only one or two facilities of each type.

A complete description of the tested facilities, testing procedures, and emissions data used in this report is found in a publicly released draft of "Municipal Waste Combustion Study: Emissions Data Base for Municipal Waste Combustors."[1]

3.2.1 Emission Factors

Based on data from emission tests, emission factors were developed to represent emissions typical of municipal waste combustors. In general, emission factors were derived by averaging available emissions data for a given pollutant and facility type. Emission factors were developed for each of the pollutants analyzed in this risk assessment including the following organic pollutants (or classes of organic pollutants): chlorinated dibenzo-p-dioxins/dibenzofurans (CDDs/CDFs), chlorophenols, chlorobenzenes, formaldehyde, polychlorinated biphenyls (PCB), and benzo-a-pyrene [B(a)P]. Emission factors also were developed for the following inorganic pollutants: arsenic, beryllium, mercury, lead, cadmium, hexavalent chromium, and hydrogen chloride. Different emission factors were developed for each pollutant to represent emissions from existing and projected municipal waste combustor populations.

In deriving emission factors a dichotomy was made between existing and projected municipal waste combustors. This dichotomy was based on the premise that municipal waste combustors currently marketed represent distinct improvements in design, combustion efficiency and pollution control when compared to municipal waste combustors planned and built just a decade ago. Subjective judgments were made regarding the selection of certain facilities to represent potential pollutant emissions from existing and projected municipal waste combustors.

The difference in emission factors for existing and projected municipal waste combustors reflect the discriminating use of different emissions test data. In the case of existing facilities, emission factors were derived using emissions data from most tested facilities with the exception of specific state-of-the-art technologies. By contrast, only data from facilities judged to have combustion technologies representative of newer, more advanced technologies were used to derive emission factors for projected facilities. Projected facilities were also assumed to use good combustion controls for minimizing the formation and emission of organic pollutants.

The following assumptions were made regarding baseline control of pollutant emissions. For the existing combustor population, the baseline scenario assumed uniform use of electrostatic precipitators (ESPs) because available emissions data were collected from ESP-controlled units. A variety of PM control devices are actually in place on existing units. Baseline conditions assumed 20 percent control of organic pollutants and current levels of particulate matter control.[2] Control levels for metal emissions (except mercury) were assumed to be proportional to control levels for PM emissions.

For the projected population, baseline conditions assumed the uniform applications of ESPs, although some States are requiring more stringent control. Organic pollutant and PM emissions (with the exception of mercury) were assumed to be controlled by 20 percent and 99 percent, respectively, and metal emissions were assumed to be controlled in proportion to particulate emissions. These control efficiencies reflect optimized control of these pollutants using these control technologies. No control of inorganic acid gases such as sulfur dioxide or hydrogen chloride was assumed for baseline conditions of either existing or projected facilities. The assumptions used to define the control scenarios considered in this analysis are summarized in Table 3-3.

The method used to derive the emission factors for the various pollutant classes are presented in the following subsections.

TABLE 3-3. CONTROL SCENARIOS MODELED FOR EXISTING AND PROJECTED MUNICIPAL WASTE COMBUSTORS

Population/ Control Scenario	Control Technology	Control Efficiency (%)			
		Organics	Metals (except Hg)	Hg	HCl
Existing					
Baseline	ESP	20	*	30	0
Controlled	Dry alkaline scrubber/ fabric filter	95	99.5	50	90
Projected					
Baseline	ESP	20	99	30	0
Controlled	Dry alkaline scrubber/ fabric filter	95	99.5	50	90

[*] For existing municipal waste combustors, baseline control levels for metal emissions (except Hg) were assumed to be proportional to actual control levels achieved by existing particulate matter controls, as measured during emission tests.

3.2.1.1 Organic Emission Factors. Organic pollutant emission factors, expressed in grams of pollutant per megagram of waste combusted, are presented in Table 3-4 for the different municipal waste combustion technologies modeled. The organic emission factors were developed by calculating for each tested facility the measured organic pollutant emission rate and then dividing by the waste combustion rate. For facilities currently equipped with control devices other than ESPs (i.e., dry alkaline scrubbers in combination with fabric filters), the measured organic pollutant emission rate was first adjusted to reflect control levels achieved by ESPs. Emission factors for each municipal waste combustion technology modeled were determined by calculating the average emission rate (in g/Mg of waste combusted) for all facilities of a given type.

The facilities providing test data used in the calculation of individual organic emission factors are listed in Table 3-5. As the table shows, for some organic pollutants there were very few test data. Also observed from Table 3-5, data from different facilities were used in determining emission factors for existing vs. projected municipal waste combustors. Efforts were made to use the most appropriate data to represent emissions from different types of combustors and for different scenarios. Some of the existing facilities, for example Hampton (massburn), may not represent performance levels expected of new facilities. Test data from this facility were, therefore, excluded when computing average emission factors for CDD and CDF for the projected population of massburn heat recovery facilities. However, for some other organic pollutants, for example, B(a)P, there were not state-of-the-art data available, so the data from Hampton were used, even though they may not be considered representative of new facilities.

Most of the tested facilities listed in Table 3-5 are equipped with particulate controls only. Although little data are available on the performance of particulate controls in controlling organics, preliminary results from recent tests on ESPs indicate a range of 0 to 50 percent control of CDDs and CDFs may be occurring,[3,4] and data from earlier tests at the Chicago NW facility indicated 0-45 percent control of other organic compounds by an ESP.[5] Based on these data, 20 percent control of organics was assumed to be achieved by ESPs alone without appreciable prior cooling of the flue

TABLE 3-4. ORGANIC EMISSION FACTORS FOR MUNICIPAL WASTE COMBUSTORS (g/Mg of waste combusted)[a]

Organic	Facility Type	Emission Factor	
		Existing	Projected[b]
CDD/CDF[c,d]	Massburn-Heat Recovery	0.00096	0.000058
	Massburn-Nonheat Recovery	0.0031	N/A[e]
	RDF	0.002	0.0012
	Modular	0.000069	0.000069
Benzo-a-pyrene [B(a)P]	Massburn-Heat Recovery	0.048	0.048
	Massburn-Nonheat Recovery	0.048	N/A[e]
	RDF	0.13	0.13
	Modular	0.0038	0.0038
Polychlorinated Biphenyls (PCB)	Massburn-Heat Recovery	0.0021	0.00021
	Massburn-Nonheat Recovery	0.0021	N/A[e]
	RDF	1.4	1.4
	Modular	0.0045	0.0045
Formaldehyde	Massburn-Heat Recovery	5.2	2.5
	Massburn-Nonheat Recovery	5.2	N/A[e]
	RDF	1.1	1.1
	Modular	1.3	1.3
Chlorobenzenes	Massburn-Heat Recovery	0.32	0.0089
	Massburn-Nonheat Recovery	0.32	N/A[e]
	RDF	0.14	0.14
	Modular	0.018	0.018
Chlorophenols	Massburn-Heat Recovery	0.46	0.018
	Massburn-Nonheat Recovery	0.46	N/A[e]
	RDF	0.18	0.18
	Modular	0.018	0.018

[a] Emission factors were derived from an average of available emissions data for a given pollutant and facility type.

[b] Data used to calculate emission factors for projected facilities exclude test data for facilities not equipped with state-of-the-art combustion technology.

[c] CDD/CDF refers to the class of 75 chlorinated dibenzo-p-dioxins (CDD) and 135 chlorinated dibenzofurans (CDF) compounds.

[d] CDD/CDF emissions are reported as the 2,3,7,8-TCDD equivalent concentration based on homologue-specific data.

[e] N/A = Not Applicable. All projected municipal waste combustors are assumed to be heat recovery facilities.

TABLE 3-5. MUNICIPAL WASTE COMBUSTION FACILITIES FROM WHICH TEST DATA WERE USED TO DERIVE ORGANIC POLLUTANT EMISSION FACTORS

#	MWC Tested	CDD/CDF	B(a)P	PCB	Formaldehyde	Chlorobenzenes	Chlorophenols
EXISTING MWC							
1	Massburn Non Heat Recovery	Philadelphia NW	Hampton	Hampton	Hampton Peekskill	Chicago NW Hampton	Chicago NW Hampton
6	Massburn Heat Recovery	Quebec Saugus Chicago Peekskill Hampton N. Andover	Hampton	Chicago NW Hampton	Hampton Peekskill	Chicago NW Hampton	Chicago NW Hampton
6	Modular	PEI, Dyersburg N. Little Rock Mayport, Oneida Cattaraugus	Cattaraugus	PEI Cattaraugus	Dyersburg Cattaraugus	PEI	PEI
5	RDF	SWARU Akron Wright Patterson Albany Niagara Falls	Albany	Albany SWARU	Albany Akron Niagara Falls	Wright Patterson SWARU	Wright Patterson SWARU
PROJECTED MWC							
7	Massburn	Wurzburg N. Andover Saugus Peekskill Tulsa Marion Co. Chicago NW	Hampton	Chicago NW	Peekskill	Chicago NW	Chicago NW
5	RDF	SWARU[a] Akron Wright Patterson Albany Niagara Falls	Albany	Albany SWARU[a]	Albany Akron Niagara Falls	Wright Patterson SWARU[a]	Wright Patterson SWARU
6	Modular	PEI, Dyersburg N. Little Rock Mayport, Oneida Cattaraugus	Cattaraugus	PEI Cattaraugus	Dyersburg Cattaraugus	PEI	PEI

[a] Adjusted to reflect judgment about recent modifications.

gas. In calculating the baseline emission factors, representing the use of ESPs only (Table 3-4), organic emissions measurements for the small number of facilities achieving greater control of organics with dry alkaline scrubbers in combination with particulate matter control devices were adjusted to reflect 20 percent control achievable by ESPs.

The organic emission factors in Table 3-4 were applied directly to estimate organic emissions from municipal waste combustors under the baseline control scenario. Under the controlled scenario, 95 percent control of organic emissions was assumed to be achievable through uniform application of dry alkaline scrubbers combined with fabric filters. This level of control is supported by recent emission tests conducted by both Environment Canada and the EPA.[6] Full-scale and pilot-scale systems equipped with dry scrubbers in combination with a baghouse system (fabric filters) generally achieved better than 95 percent control of organic emissions when operated at temperatures below 200°C. To estimate organic emissions after application of dry alkaline scrubbers in combination with fabric filters, the emission factors in Table 3-4 were adjusted as follows:

$$EF_C = EF_B \frac{(1 - 0.95)}{(1 - 0.20)}$$

where:

EF_C = emission factor for controlled scenario
EF_B = emission factor for baseline control scenario
0.95 = organic control efficiency for alkaline scrubber
0.20 = organic control efficiency for ESP

3.2.1.2 Metal Emission Factors. Table 3-6 displays metal emission factors estimated for the various municipal waste combustion technologies. Metal emission factors are expressed in units of micrograms of pollutant per gram of controlled particulate emissions. The specific municipal waste combustion facilities from which test data were used to calculate the individual metal emission factors are listed in Table 3-7. The test data shown were used to calculate emission factors representing both the existing and projected municipal waste combustor populations.

TABLE 3-6. METAL EMISSION FACTORS FOR MUNICIPAL WASTE COMBUSTORS
(ug metal/g controlled particulate emissions)

Metal	Facility Type	Emission Factor
Arsenic	MB Water Wall	335
	MB Refractory	11.9
	RDF	90
	Modular	353
Beryllium	MB Water Wall	0.091
	MB Refractory	11.9
	RDF	65
	Modular	0.036
Cadmium	MB Water Wall	686
	MB Refractory	1,228
	RDF	106
	Modular	3,310
Chromium (+6)	MB Water Wall	9,476
	MB Refractory	415
	RDF	10,912
	Modular	818
Lead	MB Water Wall	13,425
	MB Refractory	61,105
	RDF	10,530
	Modular	72,500
Mercury	MB Water Wall	8,163
	MB Refractory	6,770
	RDF	868
	Modular	2,340
Nickel	MB Water Wall	6,855
	MB Refractory	2,812
	RDF	5,770
	Modular	1,610

TABLE 3-7. MUNICIPAL WASTE COMBUSTION FACILITIES FROM WHICH TEST DATA WERE USED TO ESTIMATE METALS EMISSION FACTORS

Pollutant Metals	Facility Type	MWC Facilities Tested
Arsenic	MB Water Wall	Baltimore, Braintree, Hampton, Munich, Wurzburg
	MB Refractory	Tsushima
	RDF	Akron, Albany
	Modular	Tuscaloosa, Dyersburg, Prince Edward Island
Beryllium	MB Water Wall	Braintree, Hampton, Tulsa, Munich
	MB Refractory	Tsushima
	RDF	Albany
	Modular	Dyersburg
Cadmium	MB Water Wall	Braintree, Hampton, Munich, Malmo, Wurzburg
	MB Refractory	Washington, Alexandria, Nicosia, Tsushima
	RDF	Albany
	Modular	Dyersburg, Prince Edward Island
Chromium	MB Water Wall	Baltimore, Braintree, Hampton, Munich, Wurzburg
	MB Refractory	Washington, Alexandria, Nicosia, Tsushima
	RDF	Akron, Albany
	Modular	Dyersburg, Prince Edward Island
Lead	MB Water Wall	Braintree, Hampton, Tulsa, Munich, Malmo, Wurzburg
	MB Refractory	Washington, Alexandria, Nicosia, Tsushima
	RDF	Akron, Albany
	Modular	Dyersburg, Prince Edward Island
Mercury	MB Water Wall	Braintree, Hampton, Tulsa, Malmo
	MB Refractory	Tsushima
	RDF	Akron, Albany
	Modular	Dyersburg, Prince Edward Island
Nickel	MB Water Wall	Hampton, Munich, Wurzburg
	MB Refractory	Washington, Alexandria, Nicosia, Tsushima
	RDF	Akron, Albany
	Modular	Dyersburg, Prince Edward Island

Metal emission factors were derived by first calculating for each tested facility the ratio of metals to particulate emissions in controlled particulate emission measurements. The average metal to particulate ratio was calculated from the available test data for each combustor technology to determine baseline emission factors for these metals. The metal emission factors in Table 3-6 reflect the current control levels achieved by the particulate control devices in place at each of the tested facilities. In general, existing particulate controls may be achieving about 99 percent control of particulate and metals emissions, except mercury emissions which are assumed in this analysis to be controlled only about 30 percent by electrostatic precipitators at temperatures above 200°C.

To estimate metal emissions from municipal waste combustors under the baseline scenario, the emission factors in Table 3-6 were directly applied. To generate estimates of emissions under the controlled scenario, fabric filters were assumed to be capable of achieving 99.5 percent control of metals emissions except for mercury emissions. Mercury emissions were assumed to be only controlled 50 percent. However, tests conducted by Environment Canada on a pilot-scale dry alkaline scrubber combined with fabric filters demonstrated that mercury could be controlled by as much as 90 percent if the devices were operated at temperatures at or below 145°C.[7] To estimate metal emissions from municipal waste combustors under the controlled scenario, the emission factors in Table 3-6 for all metal pollutants except mercury were adjusted as follows:

For existing facilities:

$$EF_C = EF_B \frac{1 - 0.995}{1 - CE_B}$$

For projected facilities:

$$EF_C = EF_B \frac{1 - 0.995}{1 - 0.99}$$

where:

EF_C = emission factor for controlled scenario
EF_B = emission factor for baseline scenario
0.995 = metal control efficiency achieved with particulate controls under the controlled scenario

0.99 = metal control efficiency achieved with particulate controls for projected facilities under baseline scenario

CE_B = metal control efficiency achieved with existing particulate controls for existing facilities under baseline scenario

The mercury emission factors, based on estimated controlled particulate emissions, were adjusted as follows:

$$EF_C = EF_B \frac{1 - 0.50}{1 - 0.30}$$

where:

EF_C = emission factor for controlled scenario

EF_B = emission factor for baseline scenario

0.50 = mercury control efficiency achieved with particulate controls under the controlled scenario

0.30 = mercury control efficiency achieved with existing particulate controls

3.2.1.3 Hydrogen Chloride (HCl) Emission Factors. Baseline HCl emission factors are presented in Table 3-8 for the types of municipal waste combustors modeled. Emission factors are expressed in units of grams of HCl per megagram of waste combusted. To derive this, HCl emission rates were divided by the waste combustion throughput for each tested facility. Two sets of emission factors were estimated from these data: average, and maximum to be used to model long-term and short-term averages. Average emission factors were determined by calculating the average of all HCl measurements for a given combustor type. Maximum emission factors represent the highest reported value of HCl emissions per quantity of waste combusted for each combustor type. It should be noted that because the test data are themselves averages of several runs, generally of 3 to 4 hours duration, this may be a conservative maximum, and short-term average.

TABLE 3-8. HCl EMISSION FACTORS FOR MUNICIPAL WASTE COMBUSTORS (kg/Mg of waste combusted)[a]

Facility Type	HCl Emission Factor	
	Average	Maximum
Massburn-Heat Recovery	4.13	7.15
Massburn-Nonheat Recovery	4.13	7.15
RDF	2.57	N/A[b]
Modular	3.58	4.43

[a] kg/Mg = kilogram of HCl emitted per megagram (metric ton) of waste combusted.

[b] Not available.

The baseline emission factors in Table 3-8 reflect zero percent control of HCl emissions without the use of dry alkaline scrubbers. These emission factors were used to estimate annual average and maximum HCl emissions from existing and projected municipal waste combustors under the baseline control scenario. Ninety percent control of HCl emissions by dry alkaline scrubbers is assumed under the controlled scenario (i.e., reflecting use of dry alkaline scrubbers in combination with fabric filters). HCl emissions under the controlled scenario were estimated by adjusting the emission factors in Table 3-8 as follows:

$$EF_C = EF_B\ (1 - 0.9)$$

where:

EF_C = emission factor for controlled scenario
EF_B = emission factor for baseline scenario
0.9 = HCl control efficiency achievable under controlled scenario

3.2.1.4 Uncertainties and Assumptions in Emission Estimates. Emissions estimates used in the risk assessment are based on emissions test data collected from a limited number of tests on municipal waste combustors. The tested facilities vary widely in design and operational performance. In addition, the tests were conducted with different objectives with the result that testing protocols and the level of detail of reported data was variable. Finally, specific sampling and analysis methods were not described in detail and may not be consistent for all tests. Direct comparison of the test results for different facilities is therefore difficult. As described above, average emission factors were developed from the test data to represent typical emissions from existing and projected municipal waste combustors.

The following uncertainties are noted. First, the emissions data used to calculate emission factors represent results of tests on emissions from municipal waste combustor stacks only. Fugitive emissions were not measured and are not included in the emission factors.

Secondly, the emissions data came from regulatory compliance tests conducted over a relatively short period of time, and may not be representative of long-term emissions over the 30-year life of a typical municipal waste combustor. Periods of downtime and diminished performance of pollution control devices over time would contribute to variations in actual emissions versus tested emissions for a given facility.

Third, emissions data were taken from only a small number of the existing municipal waste combustor facilities and, therefore, may not be statistically representative of the industry as a whole in all cases. For example, CDD/CDF emission estimates from massburn facilities without heat recovery were based on tests from only one facility, the Philadelphia NW facility. Emissions for this facility may or may not be representative of those from other nonheat recovery massburn facilities. However, facilities without heat recovery are generally older than facilities with heat recovery, so there may be a real basis for the difference in estimated CDD/CDF emissions.

Similarly, the emission estimates for existing massburn energy recovery facilities are influenced by emission levels found at the Hampton facility, which are much higher than those found at other tested massburn facilities with heat recovery. Without further tests, one cannot determine the extent to which other facilities may or may not have emissions comparable to Hampton. The emissions from Hampton were excluded in calculating the average emission factor for projected facilities with the assumption that the distinct operational improvements in design and operation at new facilities may make Hampton emissions non-representative of projected facilities.

In another case, emission estimates for projected RDF facilities are influenced by the inclusion of emission data from the SWARU RDF facility (Hamilton, Ontario). Emissions of CDDs/CDFs are significantly higher at SWARU than those of other tested RDF facilities, and SWARU had known operational problems at the time the tests were performed (which recent modifications have attempted to correct).[8] There are insufficient data for RDF facilities, however, to statistically determine whether these known problems are unique to SWARU, and, therefore would make SWARU emission data unsuitable for use in estimating emissions from new RDF facilities. To account for recent design and operational improvements at SWARU, the measured emissions of CDDs/CDFs were reduced 50 percent when included in the emission factor derivation for projected RDF facilities.

Some uncertainties are associated with emissions estimates for CDDs/CDFs. For example, recovery of CDD/CDF from the Modified Method 5 stack sampling trains in some cases has been reported to be as low as 10 percent. Thus, actual emission levels could be higher than reported by an order of magnitude. However, recovery has also been reported to be significantly higher. The ranges of cancer risk results presented in Section 2.3.1 account for this uncertainty.

The ranges do not account, however, for the fact that a significant portion (80%) of the organic pollutants emitted from the stacks of municipal waste combustors have not been fully identified and quantified. Although some portion of the mixture may be carcinogenic, the carcinogenic potential is unknown.

In the case of metals, in many tests only the particle phase was analyzed for metals, while in a few tests metals were measured in both the particle and vapor phase. Measurements of metals in both phases may result in higher emission estimates. No attempt was made to account for this uncertainty in constructing ranges of cancer risk results.

3.2.2 Particle Size Distribution for Deposition Modeling

A limited amount of data are available on particle size fractionalization of particulate matter entrained in municipal waste combustor emissions. A series of stack tests conducted by EPA at the municipal waste combustor in Braintree, Massachusetts provided the data on a likely distribution of particulate matter by particle size used in the indirect exposure analysis.[9] The Braintree municipal waste combustor was tested at relatively high particulate emission rates (6.7 kg/hr), in exceedance of EPA requirements, and the particulate matter control efficiency of the electrostatic precipitator was calculated to be only 74 percent. Total particulate concentrations averaged 215 mg/dscm at 12 percent CO_2. The particle size distribution measured at this facility may or may not be representative of state-of-the-art equipment, but it should provide reasonably conservative, tending toward the worst-case, data for estimating the surface deposition of particles in municipal waste combustor emissions.

Table 3-9 displays the average particle-size distribution measured at the Braintree municipal waste facility. Recently, tests have been conducted at the outlet of the fabric filters operating on a massburn, heat recovery municipal waste combustor in West Germany.[10] Table 3-10 summarizes the size distribution of particulate matter at the Wurzburg, West Germany, massburn heat recovery municipal waste combustor. In comparison to the Braintree municipal waste combustor data, the emission rates of particulate matter at the outlet of the fabric filters averaged 9.0 mg/dscm (at 12 percent CO_2) over 3 test runs. Approximately one-half of the particulate matter emissions at Wurzburg were less than 0.5 microns in diameter although, over one-third of the emissions were of a diameter greater than 12 microns.

For purposes of estimating deposition of individual pollutants, estimations of the relationship between the pollutant concentration and particle size were assumed. For inorganic pollutants, data from a limited number of municipal waste combustors have been evaluated to determine the affinity of individual metals for adsorption onto surfaces of various particle diameters.[11] In general, the maximum mass of metals emissions was found to be associated with fine particles, e.g., two microns or less in aerodynamic diameter. Table 3-11 is a summary of the particle phase distribution of metallic elements adsorbed on particulate matter.

Data on the distribution of organic compounds adsorbed onto variously sized particulates was not available. The lack of data on the relationships between particle size and organic compound concentration was overcome by assuming that compounds are distributed in proportion to the percent of total particle surface area available for adsorption for each particle size category (assuming particles are perfect spheres). The mass emission rate of the organic pollutant was distributed by particle size by computing proportion and available surface area for a given particle size,[12] and holding particle density constant. The particle weight is proportional to volume if density is constant. Therefore, the ratio of the surface area to volume is proportional to the ratio of surface area to weight for a particle with a given radius. Multiplying this proportion times the weight fraction of particles of a specific diameter yielded an estimate of the amount of surface area available for chemical adsorption.

TABLE 3-9. PARTICLE-SIZE DISTRIBUTION DETERMINED IN PARTICULATE MATTER DISTRIBUTIONS AT THE BRAINTREE MUNICIPAL WASTE COMBUSTOR[a,b,c]

Geometric Mean Particle Diameter (Microns)	Percent of Total Particle Mass
>15.0	12.8
12.5	10.5
8.1	10.4
5.5	7.3
3.6	10.3
2.0	10.5
1.1	8.2
0.7	7.6
<0.7	22.4

[a] Data is an average of 6 stack test runs.

[b] The facility is equipped with an electrostatic precipitator rated at an average particulate removal efficiency of 74 percent at time of test.

[c] The facility recovers steam from the combustion of about 120 metric tons of municipal waste per day.

Source: Reference 9

TABLE 3-10. TYPICAL PARTICLE-SIZE DISTRIBUTION DETERMINED IN PARTICULATE EMISSIONS AT THE WURZBURG MASSBURN MUNICIPAL WASTE COMBUSTOR[a,b,c]

Particle Diameter (Microns)	Percent of Particle Mass
>12.0	37.8
7.5 - 12	0.0
5.1 - 7.5	1.0
3.5 - 5.1	1.5
2.3 - 3.5	3.6
1.1 - 2.3	2.0
0.7 - 1.1	0.5
0.47 - 0.7	1.0
<0.47	52.6

[a] Data represents one test day, but is typical of performance.

[b] The facility is equipped with a dry venturi scrubber coupled with fabric filters for air pollution control.

[c] The facility has a capacity of 600 metric tons municipal waste per day.

Source: Reference 10

TABLE 3-11. RATIO OF METAL EMISSIONS AS A FUNCTION OF PARTICLE DIAMETER (>2u/<2u)

Pollutant	Ratio
Arsenic	25/75
Beryllium	34/66
Cadmium	34/66
Chromium	41/59
Lead	12/88
Nickel	41/59

Source: Reference 11

Three particle size categories were selected to characterize the data: greater than 10 microns, 2 to 10 microns, and less than 2 microns. The fraction of total surface area available for chemical adsorption was calculated to be 0.03, 0.095, and 0.875, respectively. The specific emission rate for each organic pollutant considered in the analysis was multiplied by the fraction of available surface area to estimate the pollutant emission rate corresponding to the particle size distribution. For example, if the emission rate of formaldehyde was estimated to be 80 milligrams per second of plant operation, then the formaldehyde emission rate in consideration of available surface area would be: (0.030 x 80 mg/s) = 2.4 mg/s for particles greater than 10 microns; (0.095 x 80 mg/s) = 7.6 mg/s for particle sizes ranging from 2 to 10 microns; (0.875 x 80 mg/s) = 70 mg/s for particles less than or equal to 2 microns in diameter. The organic pollutants emissions were determined on this basis for purposes of wet and dry surface deposition.

To calculate emission rates of inorganic compounds corresponding to the particle size distribution, ratios of mass of the compound per particle size category were assumed. For example, Table 3-11 indicates that approximately 25 percent of arsenic has been found adsorbed to particles greater than 2 microns in diameter, and thus the ratio is 25 percent/75 percent for >2/<2 microns. If arsenic were emitted at a rate of 2.3×10^{-3} grams per second, then the emission rate per particle size cutoff, taking into consideration available surface area for adsorption, would be 1.4×10^{-4} g/s for particles greater than 10 microns; 4.4×10^{-4} g/s for particles 2 to 10 microns, and 1.7×10^{-3} g/s for particles less than 2 microns.

The particle size and surface area distribution were kept constant in the analysis of inorganic and organic compound emissions resulting from the application of two distinct sets of air pollution control systems: ESPs and the combination dry venturi scrubber and fabric filters. Comparisons between the particle size distributions depicted in Tables 3-7 and 3-8 do show distinct differences between emissions resulting from ESP control or fabric filter control. However, it was felt that not enough particle mass fractionalization data exists to generalize a distribution pattern for each control device, based on these limited measurements. Therefore, EPA has assumed, for purposes of estimating deposition of specific pollutants, a unitized particle size distribution of both pollution control systems.

3.3 ESTIMATING EXPOSURE AND EVALUATING EFFECTS

3.3.1 Direct Exposure to Municipal Waste Combustor Emissions

Direct inhalation exposure to emissions from municipal waste combustors was evaluated using the Human Exposure Model (HEM). Based on the exposure levels predicted by the HEM, an estimate of cancer risk, noncancer risk, and the potential for environmental effects resulting from direct emissions from municipal waste combustors was made. The pollutants considered in each part of this analysis are listed in Table 3-12. The key elements of the analysis of exposure and effects associated with direct emissions from municipal waste combustors are described below.

3.3.1.1 Human Exposure Model. The HEM is a general model designed to estimate the population exposed to air pollutants emitted from stationary sources. The HEM is comprised of an atmospheric dispersion model with included meteorological data and a population distribution based on Bureau of Census data. In addition, the HEM contains a procedure for estimating cancer risks due to the predicted exposure. Based on source data, the model estimates the magnitude and distribution of ambient air concentrations of the pollutant in the vicinity of the source. These concentration estimates are coupled with the population data to estimate public exposure to the pollutant. A more detailed description of the HEM is included in Appendix A.

Exposure is the product of the population and the concentration to which the population is exposed. To form this product, both the concentration and the population must be known at the same location or point. The HEM uses the latitude and longitude of the facility to determine the population of the study area. The population data base is comprised of the 1980 Census Data Base broken down by block group/enumeration district (BG/ED). The population data base contains the population centroid coordinates (latitude and longitude) and the 1980 population of each BG/ED in the country (about 300,000 centroids in 50 states plus the District of Columbia). A population centroid is the population-weighted geographical center of a BG/ED for known geodetic coordinates. For BG/ED centroids located between 0.2 km and 3.5 km from the facility, populations are apportioned among neighboring polar grid points. A

TABLE 3-12. POLLUTANTS CONSIDERED IN ANALYSIS OF DIRECT EMISSIONS FROM MUNICIPAL WASTE COMBUSTORS

Analysis	Pollutants
Cancer Risk	Organic
	CDD/CDF[a] chlorophenols chlorobenzenes formaldehyde polychlorinated biphenyls (PCB) polycyclic aromatic hydrocarbons (PAH)[b]
	Inorganic
	arsenic beryllium cadmium chromium +6
Noncancer Risk	Lead
	Mercury
	HCl
Environmental Effects	HCl

[a] CDD/CDF refers to 75 chlorinated dibenzo-p-dioxin compounds and 135 chlorinated dibenzofuran compounds.

[b] PAH compounds are represented by benzo(a)pyrene in this analysis.

polar grid point is one of the 160 receptors at which concentrations are estimated by the dispersion model. There are 64 (4 x 16) polar grid points within this range. Both concentration and population counts are thus available for each polar grid point. Log-log linear interpolation is used to estimate the concentration of each ED/BG population centroid located between 3.5 km and 50 km from the source. Concentration estimates for 96 (6 x 16) grid points (receptors at 5, 10, 20, 30, 40, and 50 km from the source along each of the 16 wind directions) resulting from dispersion modeling are used as reference points in the interpolation.

Other information needed by the HEM to predict exposure include the vent height, vent diameter, gas discharge temperature, gas emission velocity, and building cross-sectional area for each vent modeled. The emission rate of each pollutant for which exposure estimates are needed must also be specified.

For this analysis, ambient concentrations and exposure were modeled using the HEM around every municipal waste combustor currently known by EPA to be operating in the United States. Ambient concentrations and exposures were modeled under both control scenarios. This modeling was performed based on site-specific meteorology, population, and emissions data. Emission rates of organics and metals for individual facilities were specified based on the appropriate baseline and controlled emission factors (see Section 3.2.1) scaled according to the design waste throughput or controlled particulate emission rate of each facility. For projected municipal waste combustors, ambient concentrations and exposure under both control scenarios were modeled using the HEM for three facilities representing each of the three combustor types: a 1,000 ton/day massburn, a 1,500 ton/day RDF, and a 250 ton/day modular combustor. The emission factors developed to represent emissions from projected municipal waste combustors (see Section 3.2.1) were multiplied times the model plant waste combustion capacity or controlled particulate emission rate to generate emission rates. Meteorology and population data for modeling these three facilities were selected by assuming the following locations: downtown and suburban sites in Boston, Massachusetts, and Los Angeles, California, for the massburn and RDF facilities; and downtown and suburban sites in Erie County, New York, and Ukiah, California, for the modular combustor. These locations were selected to represent places where these types of combustors are expected to be located. Stack parameter data used in modeling both the existing and projected combustors are given in "Municipal Waste Combustion Study: Characterization of the Municipal Waste Combustion Industry."[13]

The HEM employs a number of simplifying assumptions when computing exposures to a pollutant, including:

1. It is assumed that most exposure occurs at population weighted centers (centroids) of block group and enumeration districts (BG/ED) because the locations of actual residences are not contained in available databases. The model relies on information provided in a database developed by the U.S. Census Bureau.
2. It is assumed that people reside at these centroids for their entire lifetimes (assumed to be 70 years for calculating cancer risk).
3. It is assumed that indoor concentrations are the same as outdoor concentrations.
4. It is assumed that plants emit pollutants at the same emission rate for 70 years. Long-term emission rates are not known.
5. It is assumed that the only source of exposure is the ambient air and resuspension of pollutants _via_ dust is not considered.
6. It is assumed that there is no population migration or growth.
7. The model does not provide for discriminating exposure situations that may differ with age, sex, health status, or other situations. Susceptible population subgroups are not considered.

3.3.1.2 _Direct Inhalation Nationwide Cancer Risk Evaluation_. The exposure estimates predicted by the HEM are combined with measures of carcinogenic potency to estimate the probability of cancer by direct inhalation of municipal waste combustor emissions. The estimation of carcinogenic potency is expressed as the "unit risk." The unit risk estimate for an air pollutant is defined as the probability of the upper 95 percent confidence limit of the lifetime cancer risk that could occur in a population in which all individuals are exposed continuously from birth throughout their lifetimes (70 years) to a unit concentration (e.g., 1 ug/m^3) of the carcinogen in the air they breathe. The measures of carcinogenic potency (unit risk estimates) of pollutants emitted from municipal waste combustors were derived by EPA's Carcinogen

Assessment Group and are listed in Table 3-13. The derivation of unit risk estimates depends on a a quantitative evaluation of adverse health outcomes statistically tied to exposure to a chemical as observed in epidemiological, clinical, toxicological, and environmental research. Evidence for carcinogenic action is based on the observations of statistically significant tumor responses in specific organs or tissues. Evidence of possible carcinogenicity to humans is evaluated according to specific criteria set forth in EPA's Guidelines for Carcinogenic Risk Assessment.[14] Since risks at low exposure levels cannot be measured directly either by animal experiments or by epidemiological studies, mathematical models must be used to extrapolate from high doses to low doses characteristic of environmental exposures.

There are differences in the amount, type and quality of the data used to derive the unit risk estimates. Such differences may affect the confidence that can be assumed in the quantitative estimate of carcinogenic potency and subsequent quantification of cancer risks. The uncertainties associated with cancer risk assessment vary with the chemical. Generally, human data provide the best evidence that a chemical is a carcinogen. In the absence of human data, animal-to-man extrapolation must be performed. The unit risk estimated based on animal bioassays is considered an upper bound estimate of the excess cancer risk over a lifetime in populations exposed to the probable carcinogen. The concept of equivalent doses for humans compared to animals has little experimental verification regarding carcinogenic response and this is a major area of uncertainty. In addition, human populations are more variable than laboratory animals with respect to genetic constitution, diet, living environment and activity patterns. The overall uncertainties associated with unit risk estimates have not yet been statistically quantified. At best, the linear extrapolation model used to derive unit risk estimates provides an approximate but plausible estimate of the upper limit of risk -- it is not likely that the true risk would be much more than the estimated risk, but the true risk could very well be considerably lower. The quantitative aspects of risk assessments should not be construed as an accurate representation of the true cancer risk, but are best utilized in the regulatory-decision process, such as setting regulatory priorities based on a relative indication of the potential magnitude of population risk from chemical exposure.

TABLE 3-13. UNIT CANCER RISK ESTIMATES FOR INHALATION EXPOSURE TO SPECIFIC CHEMICALS (risk per ug pollutant/m^3 of air)

Pollutant	Unit Risk Factor	Reference
Arsenic	4.2×10^{-3}	EPA 600/8-83-021f
Benzo(a)pyrene	1.7×10^{-3}	EPA 540/1-86-022
Beryllium	2.4×10^{-3}	EPA 600/8-84-026b
Cadmium	1.8×10^{-3}	EPA 600/8-83-025f
Hexachlorobenzene	4.8×10^{-4}	EPA 600/8-84-015f
Trichlorophenol	5.7×10^{-6}	EPA 440/5-80-032
Chromium (VI)	1.2×10^{-2}	EPA 600/8-83-014f
*2378-TCDD	3.3×10^{-5} $(pg/m^3)^{-1}$	EPA 600/8-84-014f
*HexaCDD	1.3×10^{-6} $(pg/m^3)^{-1}$	EPA 600/8-84-014f
Formaldehyde	1.8×10^{-4}	Draft Document
PCB	1.2×10^{-3}	EPA 540/1-86-004

*The carcinogenic potency of TCDD and HexaCDD is such that the unit risk estimates are based on inhalation exposure to 1 picogram (10^{-12}g) per cubic meter of air.

By combining the estimates of public exposure with the unit risk estimate, the HEM produces two types of cancer risk estimates. The first, called maximum individual lifetime cancer risk, relates the risk to the individual or individuals estimated to live in the area of highest ambient air concentration as estimated by the dispersion model. These are the "most-exposed individuals" (MEIs). The second type of risk estimate, called annual cancer incidence, is a summation of all the risks to people living within 50 kilometers in all directions of the facility. Fifty kilometers is the air dispersion modeling radius used in the risk analysis. The cancer incidence is expressed as cases of cancer among all of the exposed population after 70 years of exposure; for statistical convenience, it is often divided by 70 and expressed as cancer incidences per year.

For the existing municipal waste combustor population, the annual cancer incidence estimates resulting from pollutant emissions at individual facilities were summed to represent the total nationwide cancer incidence for each combustor type under the baseline and controlled scenarios. The highest maximum individual lifetime cancer risk from direct inhalation of municipal waste combustor emissions was determined for the baseline and controlled scenarios by observing the maximum lifetime risk at each modeled facility and selecting the greatest value. The maximum individual cancer risk reduction obtainable under the controlled scenario for existing municipal waste combustors was estimated by subtracting the controlled risk estimates from the baseline risk estimates.

For the projected municipal waste combustor population, the total nationwide annual cancer incidence for each combustor technology was estimated for the baseline and controlled scenarios by multiplying the annual cancer incidence for each model plant location by the ratio of the projected total waste throughput for the specific municipal waste combustor technology category (as estimated in 1993) to the model plant capacity. As in the case of the existing municipal waste combustor population, the potential cancer risk reduction that could be achieved under the controlled scenario for projected municipal waste combustors was estimated by subtracting the difference between the controlled and baseline risk estimates.

Ranges of cancer risk resulting from direct inhalation exposure to predicted ambient concentrations, under baseline and controlled scenarios, of the carcinogens evaluated in this analysis were previously presented in Section 2.3.1. The cancer risk estimates were presented as ranges because of the uncertainties and assumptions in the analysis. As described in Section 2.3.1.1, these uncertainties include assumptions made concerning carcinogenic potency of classes of compounds, emissions estimates, and sampling and analysis methodology.

3.3.1.3 Noncancer Risk. Noncancer risk to humans exposed to direct emissions from municipal waste combustors was evaluated by comparing the maximum exposure levels predicted by the HEM to reference levels derived from consideration of noncancer health effects. This analysis was limited to a consideration of emissions of lead, mercury, and HCl.

The modeled ambient lead concentrations were compared to the National Ambient Air Quality Standard (NAAQS) for lead of 1.5 ug/m^3. NAAQS are set to protect the public health and welfare with a margin of safety. Mercury concentrations were compared to a reference level for ambient air established in review of the National Emission Standard for mercury, based on health effects and in consideration of total body burden due to ingestion and inhalation.[15]

For HCl, modeled concentrations were compared to the American Conference of Government Industrial Hygienists (ACGIH) ceiling level of 5 ppm.[16] This is a level for worker exposure established to be sufficiently low to prevent toxic injury from exposure to HCl, but on the borderline of severe irritation.

Concentrations of pollutants measured in ambient air may be contributed by multiple sources. Comparisons with modeled concentrations due to municipal waste combustors can only be done with reference to air quality in a particular location. However, in general, having modeled concentrations that exceed such reference levels indicates a possible adverse effect. Moreover, approaching such a reference level may also be indicative of adverse effects, depending on other pollutant sources in the area and general background levels.

Maximum exposure to annual average emissions of lead and mercury was estimated for every existing municipal waste combustor in the United States using the HEM. Site-specific meteorology, population, and emissions data were used in the modeling. Emission rates were determined for each facility by multiplying the emission factors times the controlled particulate emission rate or design waste throughput of the facility. The results (i.e., maximum exposure) predicted by the HEM for each individual facility were compared to the ambient concentration levels associated with noncancer effects presented above.

In the case of HCl emissions, maximum short-term (1-hr) exposures were also estimated for a 500 ton/day massburn and 108 and 200 ton/day modular combustors. For purposes of specifying meteorological and population inputs, using a short-term dispersion model, the massburn combustor was located in Oyster Bay, New York, and the modular combustors were located in Durham, New Hampshire, and in Portsmouth, New Hampshire, respectively. The source parameters for this modeling are given in Table 3-14. Emission rates were specified using the maximum HCl emission factors presented in Section 3.2.1.3. The maximum short-term HCl concentrations predicted for these three combustors were compared to the worker Threshold Limit Value (TLV) for HCl to estimate whether potential adverse health effects may result from short-term exposure to acid gases emitted from municipal waste combustors.

3.3.1.4 Welfare Effects. In addition to evaluating effects on humans from inhalation exposure to direct emissions from municipal waste combustors, potential adverse welfare effects on the environment were evaluated. Specifically, the potential for materials damage resulting from exposure to HCl emitted from municipal waste combustors was evaluated.

Modeled ambient concentrations of HCl surrounding municipal waste combustors were compared to a level estimated by the Texas Air Control Board to be associated with observed corrosion of ferrous metals.[17] The estimates were generated by modeling ambient concentrations expected to occur as a result of measured emissions from a hydrochloric acid plant. The modeled concentration at the point of observed corrosion was reported to be 3.0 ug/m^3.

To estimate the potential for HCl emissions from municipal waste combustors to cause materials damage, the exposure levels for HCl modeled for the analysis of noncancer risks (see Section 3.3.1.3) were compared to the concentration level associated with corrosion effects. This comparison was made for each individual existing combustor based on the modeled estimated

TABLE 3-14. PARAMETERS USED TO MODEL SHORT-TERM MAXIMUM HCl CONCENTRATIONS AND EXPOSURES FOR MUNICIPAL WASTE COMBUSTORS

	1	2	3
Facility type	Modular	Mass-burn	Modular
Waste capacity	108 tons/day	500 tons/day	200 tons/day
Facility location	Durham, NH	Oyster Bay, NY	Portsmouth, NH
Vent height	18.8 m	44.2 m	3.7 m
Vent diameter	1.2 m	2.1 m	0.9 m
Vent discharge velocity	11.5 m/s	17 m/s	16.25 m/s
Vent discharge temperature	455 K	350 K	437 K

annual average concentration predicted for that facility. The comparison was also made for the massburn and two modular combustors for which maximum short-term (1-hr) HCl concentrations are available.

3.3.2 Indirect Exposure from Long-Term Deposition of Municipal Waste Combustor Emissions

Long-term deposition of municipal waste combustor emissions was estimated using the Industrial Source Complex (ISC) Model. Human exposure to deposited emissions from municipal waste combustors through ingestion of contaminated soil, food, or water or through skin contact with contaminated soil were evaluated for 30- and 100-year scenarios using a series of mathematical models. These models are the Terrestrial Food Chain Model, the Surface Runoff Model, the Groundwater Infiltration Model, and the Dermal Exposure Model. The first three models were also used to evaluate levels of exposure to municipal waste combustor emissions for herbivorous animals, plants, soil and aquatic organisms. The pollutants considered in the indirect exposure analysis are listed in Table 3-15.

For evaluating the hypothesis that indirect exposure to municipal waste combustor emissions following long-term deposition may contribute appreciably to total exposure, it is not currently feasible nor practical to apply the models to every existing or planned municipal waste combustor. Therefore, EPA simplified the modeling process by selecting two representative municipal waste combustors to evaluate the potential cancer risk contribution to the "most-exposed individual" (MEI) resulting from 30- and 100-year exposure to deposited emissions from municipal waste combustors. Thirty years was chosen to represent the operating life of a typical combustor, and 100 years was selected with the assumption that once a site was dedicated to that purpose, the facility would be replaced with successive units of the same size and design. The MEI in this analysis was assumed to be a hypothetical farm family that obtained most of their food supply from the area of maximum deposition of pollutants emitted from muncipal waste combustors, and whose children ingested 0.5 grams of dirt per day. This family was assumed to live just outside the estimated boundary (200 meters, or about one-tenth of a mile) of the modeled combustion facility.

TABLE 3-15. POLLUTANTS CONSIDERED IN THE INDIRECT EXPOSURE ANALYSIS

Organics	Metals
CDD/CDF	lead
chlorobenzenes	mercury
polychlorinated biphenyls	chromium +6
formaldehyde	nickel
	beryllium

A model plant was devised to represent technologies typical of massburn municipal waste combustors currently being planned or constructed. An existing municipal waste combustion facility operating in Virginia was selected as the "model" to represent the potential adverse impacts of emissions from existing municipal waste combustor technology. The model plant configuration was selected to closely mimic actual planned massburn facilities in terms of stack parameters, building area, and pollutant emissions, but reflected the upper end of capacities currently planned. The Virginia facility was planned in the mid-1970's, and put into operation in 1980. EPA has accumulated several test reports of emissions of organic and inorganic pollutants at this facility, more so than any other incinerator site. However, this facility has known operational problems.

The methodology for modeling indirect exposure to pollutants emitted from municipal waste combustors is currently under development to reflect recent review comments received from EPA's Science Advisory Board. Because of the preliminary nature of the methodology and assumptions, EPA feels that the results should not be interpreted quantitatively at this time. Further, because the objective of the analysis was to test the hypothesis that indirect exposures could contribute significantly to the total exposure due to municipal waste combustor emissions, the analysis used conservative, potentially worst-case, assumptions. The various mathematical models used in the indirect exposure analysis are described below.

3.3.2.1 Industrial Source Complex Dispersion Model. The first step in analyzing indirect exposure to municipal waste combustor emissions was to model the deposition and ambient air concentrations of pollutants on surfaces around the Virginia and model plant facilities. This was accomplished using the Industrial Source Complex Short-Term (ISCST) Model. The ISCST Model, an extended version of the Single Source (CRSTER) Model, is designed to calculate ambient concentration or dry deposition values for time periods of 1, 2, 3, 4, 5, 8, 12, and 24 hours. If used with a year of sequential hourly meteorological data, ISCST can also calculate annual concentration or deposition values. Because the current ISCST Model has no provision for calculating wet deposition of the emissions, EPA developed a program to estimate the effect of precipitation events on the rate of surface deposition. A detailed description of the ISCST Model and the accompanying algorithm developed by EPA to predict wet deposition of pollutants is given in Appendix B.

Inputs required to run the ISCST Model include hourly meteorological data, source characteristics and receptor features. Hourly meteorological data requirements are the mean wind speed measured at height z_i, the direction toward which the wind is blowing, the wind-profile exponent, the ambient air temperature, the Pasquill stability category, the vertical potential temperature gradient and the mixing layer height. Source input data requirements include pollutant emission rates, physical stack measurements (i.e., height, diameter, base elevation, exit velocity and temperature), dimensions of nearby buildings, and gravitational settling parameters for particulate matter. Inputs on the type of receptor grid needed (i.e., polar or Cartesian) and specific receptor locations (if different from the standard receptor array) are also required.

For this analysis, preprocessed hourly meteorological data corresponding to the modeled sites were used. Emission rates modeled for each combustor are shown in Tables 3-16 and 3-17 along with other modeling parameters for the two facilities. Estimates of particle size distribution needed to model dry and wet deposition were presented in Section 3.2.2. A polar receptor grid was specified for this analysis.

The outputs for both surface deposition and ambient air impacts from the ISCST Model consisted of a concentration array for 160 receptors along each of 16 wind directions (computed every 22.5^{o} on a polar grid) and located at concentric radial distances from each facility of 0.2, 0.5, 1, 2, 5, 10, 20, 30 40, and 50 kilometers. The estimated deposition and ambient air concentrations at each receptor location was given for each modeled pollutant.

3.3.2.2 Terrestrial Food Chain Model. Humans in the vicinity of municipal waste combustors have the potential to directly ingest soil contaminated with deposited emissions, or to consume vegetation and animal tissues containing the contaminants. The Terrestrial Food Chain Model has separate components for examining each potential human exposure pathway illustrated in Figure 3-1. These components include methods for using empirical data on contaminant uptake by plant or animal tissues to estimate concentrations, and for integrating these estimates in conjunction with

TABLE 3-16. MODELING PARAMETERS FOR THE MODEL PLANT REPRESENTING PROJECTED MUNICIPAL WASTE COMBUSTORS

Massburn Municipal Waste Combustor (heat recovery)

Capacity: 3000 TPD (2727 Mg/d)

Location: Western Florida

Latitude: 27° 57

Longitude: 82° 27

Stack Height: 46 meters (m)

Stack Diameter: 3.1 m

Stack Exit Velocity: 11.3 m/s

Stack Temperature:

(a) 470° K with ESP
(b) 443° K with Dry Scrubbers/Fabric Filters (FF)

Building Configuration:

H: 35 m
W: 76 m
L: 42 m

Emission Rates (grams per second):

Organics	Baseline	Controlled
CDD/CDF	$1.50x10^{-6}$	$9.4x10^{-8}$
PCB	$6.70x10^{-6}$	$4.2x10^{-7}$
B(a)P	$1.7\ x10^{-3}$	$1.1x10^{-4}$
Chlorobenzenes	$2.8\ x10^{-4}$	$1.8x10^{-5}$
Chlorophenols	$5.68x10^{-4}$	$3.6x10^{-5}$
Formaldehyde	$8.0\ x10^{-2}$	$5.0x10^{-3}$

TABLE 3-16. (Continued)

Emission Rates (grams per second) [Continued]:

	Baseline			Controlled		
Inorganics	>10u	>2 - 10u	≤2u	>10u	>2 - 10u	≤2u
Arsenic	$1.4\text{x}10^{-4}$	$4.4\text{x}10^{-4}$	$1.7\ \text{x}10^{-3}$	$4.6\text{x}10^{-5}$	$1.5\text{x}10^{-4}$	$5.9\ \text{x}10^{-4}$
Beryllium	$5.8\text{x}10^{-6}$	$1.6\text{x}10^{-5}$	$4.4\ \text{x}10^{-5}$	$1.2\text{x}10^{-6}$	$5.8\text{x}10^{-6}$	$1.5\ \text{x}10^{-5}$
Cadmium	$1.5\text{x}10^{-3}$	$5.0\text{x}10^{-3}$	$1.2\ \text{x}10^{-2}$	$5.1\text{x}10^{-4}$	$1.6\text{x}10^{-3}$	$4.2\ \text{x}10^{-3}$
Chromium	$0.6\text{x}10^{-3}$	$1.7\text{x}10^{-3}$	$3.2\ \text{x}10^{-3}$	$2.0\text{x}10^{-4}$	$6.0\text{x}10^{-4}$	$1.1\ \text{x}10^{-3}$
Lead	$8.6\text{x}10^{-3}$	$2.8\text{x}10^{-2}$	$2.67\text{x}10^{-1}$	$2.9\text{x}10^{-3}$	$9.3\text{x}10^{-3}$	$8.9\ \text{x}10^{-2}$
Nickel	$1.5\text{x}10^{-3}$	$4.7\text{x}10^{-3}$	$8.9\ \text{x}10^{-3}$	$5.0\text{x}10^{-4}$	$1.6\text{x}10^{-3}$	$3.0\ \text{x}10^{-3}$
Mercury	$7.8\text{x}10^{-3}$	$2.5\text{x}10^{-2}$	$2.3\ \text{x}10^{-1}$	$5.6\text{x}10^{-3}$	$1.8\text{x}10^{-2}$	$1.66\text{x}10^{-1}$

TABLE 3-17. MODELING PARAMETERS FOR A MUNICIPAL WASTE COMBUSTOR IN VIRGINIA REPRESENTING EXISTING FACILITIES

Massburn Municipal Waste Combustor (heat recovery)

Capacity: 120 TPD (109 Mg/d)

Latitude: 37° 6 02"

Longitude: 76° 23 28"

Stack Height: 27.44 m (2 stacks)

Stack Diameter: 1.22 m (2 stacks)

Stack Temperature:

(a) 543° K with ESP
(b) 443° K with Dry Scrubbers/Fabric Filters

Stack Exit Velocity: 12 m/s

Building Configuration:

H: 27.28 m
W: 51.88 m
L: 59.5 m

Emission Rates (grams per second):

Organics	Baseline	Controlled
CDD/CDF	$1.40x10^{-5}$	$8.68x10^{-7}$
PCB	$1.05x10^{-5}$	$6.58x10^{-7}$
B(a)P	$1.42x10^{-4}$	$8.92x10^{-6}$
Chlorobenzenes	$5.82x10^{-4}$	$3.64x10^{-5}$
Chlorophenols	$2.10x10^{-3}$	$1.11x10^{-4}$
Formaldehyde	$6.26x10^{-3}$	$4.00x10^{-4}$

TABLE 3-17. (Continued)

Emission Rates (grams per second) [Continued]:

	Baseline			Controlled		
Inorganics	>10u	>2 - 10u	≤2u	>10u	>2 - 10u	≤2u
Arsenic	$2.4x10^{-4}$	$4.8x10^{-4}$	$2.0x10^{-3}$	$8.0x10^{-5}$	$1.8x10^{-4}$	$6.4x10^{-4}$
Beryllium	$1.6x10^{-8}$	$6.4x10^{-8}$	$1.6x10^{-7}$	$5.4x10^{-9}$	$2.2x10^{-8}$	$5.2x10^{-8}$
Cadmium	$4.8x10^{-4}$	$1.4x10^{-3}$	$4.0x10^{-3}$	$1.7x10^{-4}$	$4.4x10^{-4}$	$1.2x10^{-3}$
Chromium	$2.8x10^{-4}$	$1.1x10^{-3}$	$1.9x10^{-3}$	$9.2x10^{-4}$	$3.4x10^{-4}$	$6.4x10^{-4}$
Lead	$3.0x10^{-3}$	$9.0x10^{-3}$	$1.0x10^{-1}$	$9.6x10^{-4}$	$3.0x10^{-3}$	$3.4x10^{-2}$
Nickel	$2.6x10^{-4}$	$8.6x10^{-8}$	$1.6x10^{-3}$	$8.4x10^{-5}$	$2.8x10^{-4}$	$5.4x10^{-4}$
Mercury	$8.6x10^{-4}$	$2.8x10^{-3}$	$2.6x10^{-2}$	$1.8x10^{-4}$	$5.8x10^{-4}$	$5.2x10^{-3}$

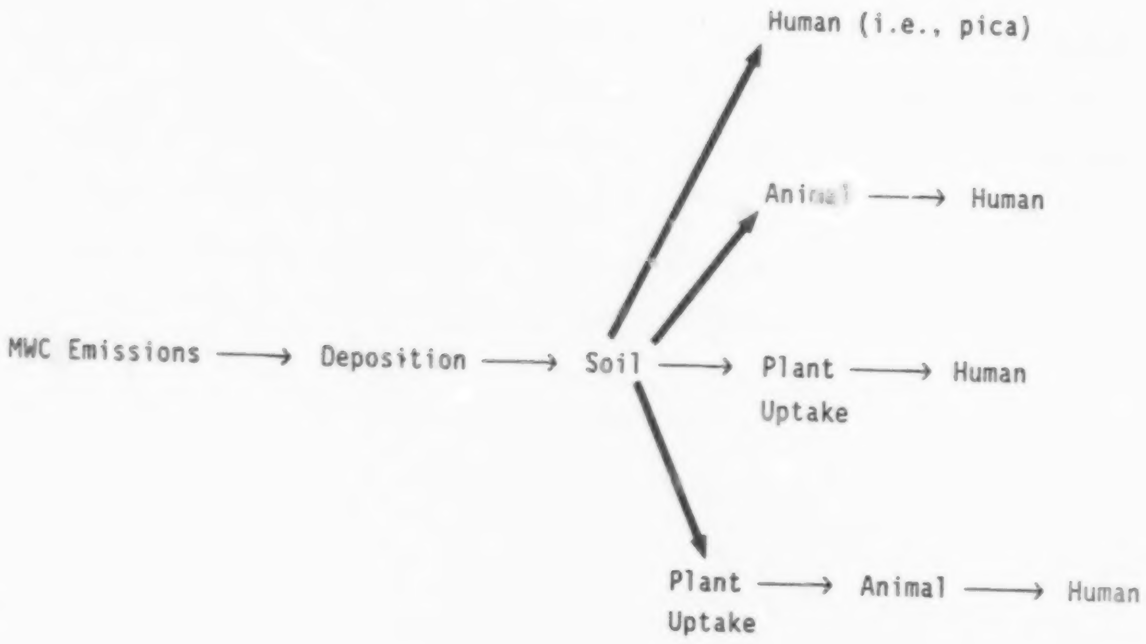
Human (i.e., pica)
Animal → Human
MWC Emissions → Deposition → Soil → Plant Uptake → Human
Plant Uptake → Animal → Human

Figure 3-1. Human Exposure Pathways Evaluated by the Terrestrial Food Chain Model

typical diet assumptions to give an estimate of potential human dietary exposure. Other components estimate potential exposure resulting from soil ingestion in children (termed, "pica"). The model predicts maximum daily intake values for the MEI based on estimates of human dietary exposure and pica in children. The MEI was assumed to be an individual residing in the area of maximum deposition of emissions. In the case of pica exposure, the MEI was assumed to be a preschool child.

In addition to evaluating human exposure to deposited emissions, the Terrestrial Food Chain Model also predicts exposure potential to herbivorous animals, plants, and soil organisms near municipal waste combustors. The ecological exposure pathways evaluated are shown in Figure 3-2. The model predicts exposure for these pathways in the same way as for human exposure except that the MEI is now a sensitive animal, plant or soil organism in the area of maximal deposition. The Terrestrial Food Chain Model is described in detail in Appendix C.

3.3.2.3 Surface Runoff Model. Following deposition, pollutants from municipal waste combustors are subject to dissolution and/or suspension in runoff after precipitation events. Runoff moves over the surface of the earth to a surface water body where it mixes with other waters. As a consequence, humans utilizing water or eating fish from the surface water body may be exposed to runoff transported contaminants. Adverse effects on aquatic life in contaminated waters may also be observed. The Surface Runoff Model includes three successive tiers of analysis beginning with simple but very conservative estimates and proceeding to more refined analyses if the first tiers predict unacceptable exposures. Both acute events and chronic exposure are evaluated using standard approaches to calculate runoff volume and associated runoff potential. The model calculates maximum receiving water concentrations of individual contaminants. In the case of human exposure, the MEI is assumed to be a person residing at the point of maximum deposition of pollutant emissions from the municipal waste combustor, and consuming water and eating fish taken from surface waters experiencing maximum contaminant concentrations from surface runoff of the deposited emissions. In the case of aquatic organism exposure, the MEI is a sensitive aquatic organism living in surface waters containing maximum contaminant concentrations. The

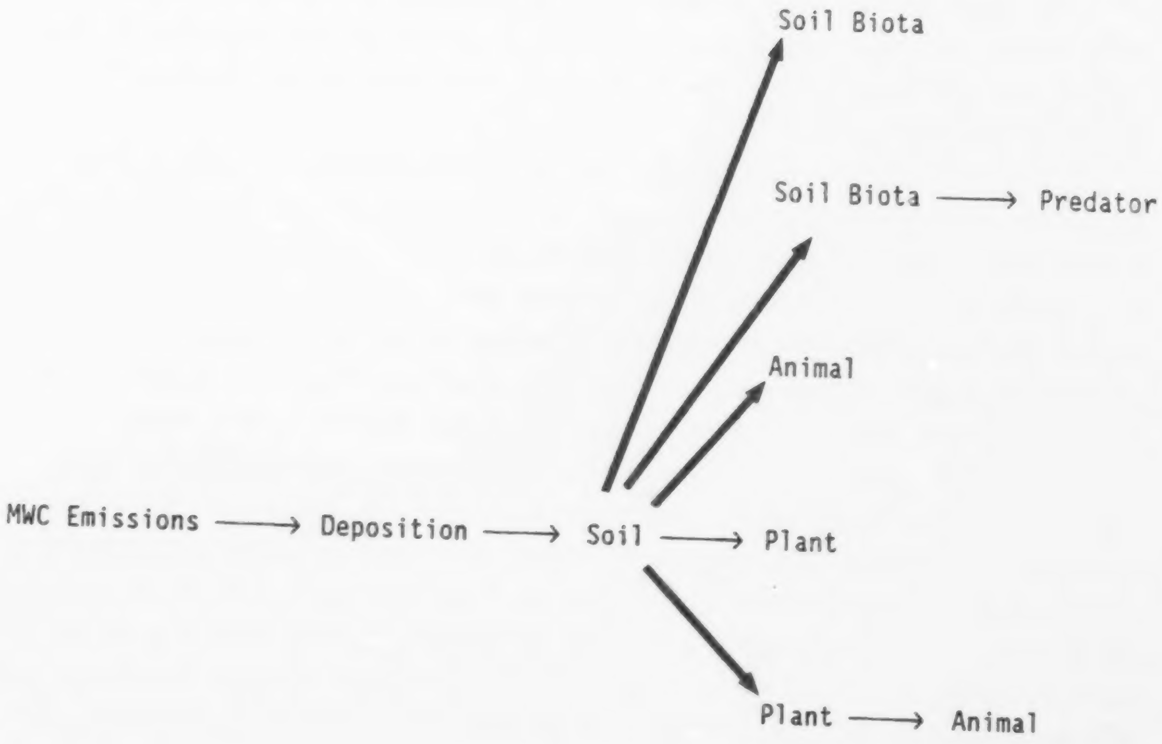
Soil Biota
Soil Biota
Predator
Animal
MWC Emissions
Deposition
Soil
Plant
Plant
Animal

Figure 3-2. Ecological Exposure Pathways Evaluated by the Terrestrial Food Chain Model

methodology in this model was originally developed to evaluate impacts from the application of municipal wastewater sludge to land. The Surface Runoff Model is described in detail in Appendix D.

3.3.2.4 Groundwater Infiltration Model. Contaminants from municipal waste combustors dissolved in rain or meltwater from precipitation events may alternatively infiltrate into the ground and enter the groundwater. As a consequence, persons using the groundwater may be exposed to groundwater transported contaminants. Aquatic life inhabiting surface water bodies fed by the contaminated aquifer could be exposed as well. The Groundwater Infiltration Model evaluates chronic exposure using standard approaches to calculate leachate generation and associated groundwater transport in the unsaturated zone. Similar to the Surface Runoff Model, the Groundwater Infiltration Model is formulated in three successive tiers which begin with simple conservative estimates and proceed to more detailed analyses if the first tiers predict unacceptable exposures. The model estimates maximum potential concentrations of contaminants at various points of the aquifer, depending on which analytical tier is being used. The human and nonhuman MEI are the same as for the Surface Runoff Model. A detailed description of the Groundwater Infiltration Model is given in Appendix E.

3.3.2.5 Dermal Exposure Model. Humans may also be exposed dermally to soils contaminated by municipal waste combustor emissions. The Dermal Exposure Model estimates a daily dermal intake of contaminants by the MEI based on the fraction of the contaminant absorbed, contact time (i.e., duration of daily exposure), exposed skin surface area, contact amount (i.e., amount of soil accumulated on skin, and soil contaminant concentration. This exposure level is then adjusted to account for the relative effectiveness of an absorbed dermal dose relative to an unabsorbed ingested dose. The MEI is assumed to be a preschool child playing in soil in the area of maximal deposition.

It should be noted that there is a fundamental lack of data for percutaneous absorption of chemicals in human skin from soil. Other factors important for estimation of human exposure to contaminants by the dermal route also have many uncertainties. In addition, systemic toxic thresholds or carcinogenic potencies of chemicals by a dermal route of exposure have not been delineated by EPA at the present time. While this model represents a possible approach for the estimation of human exposure associated with dermal exposure, it is recognized that in most, if not all cases, the available data will not provide a satisfactory basis for risk exposure calculations. The Dermal Exposure Model is fully described in Appendix F.

3.3.2.6 Evaluation of Exposure from Indirect Pathways. Because of the developmental nature of the indirect exposure methodology described previously, reliable quantitative risk estimates for indirect exposure pathways have not yet been generated. However, an analysis was performed using the preliminary methodology to predict the likelihood that indirect exposure routes could contribute significantly to the total exposure to individuals living near existing and projected municipal waste combustors, as represented by the Virginia and Western Florida model facilities. This limited analysis was performed for indirect exposure to individual pollutants through four principal routes: food ingestion, soil ingestion, water ingestion and fish ingestion. The effects of deposition of pollutants over 30 and 100 years were also considered.

Tables 3-18 and 3-19 summarize the analysis for inorganic and organic pollutants, respectively. For each pollutant evaluated using the preliminary methodology, the likelihood that a particular indirect exposure pathway could contribute significantly to total exposure is subjectively classified as either "possible" or "low likelihood." The "no data" classification was used to indicate those pollutants for which insufficient data were available to apply the preliminary methodology.

Certain aspects of the indirect exposure modeling should be considered when reviewing the qualitative observations in Tables 3-18 and 3-19. The selection criteria of both the Virginia and western Florida facilities are considered as reasonable worst-case in that the facility size, location, stack parameters, and stack pollutant emissions tended toward maximizing potential ambient impacts near each municipal waste combustor. Both facilities are located in humid climates where an excess of 40 inches of rainfall per year enhanced the effect of wet deposition of pollutants near each facility. Dryer climates would be expected to produce lower rates of wet deposition. In the western Florida case the sandy soil characteristics may have a tendency to maximize the rate of downward movement of pollutants

TABLE 3-18. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL HUMAN HEALTH EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS
(Exposure by Ingestion - Inorganic Compounds)

Facility	Control Scenario	Deposition Interval	Food Ingestion			Soil Ingestion			Surface Water Ingestion			Fish Ingestion		
			Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.		Pb, Cr Hg, Ni	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
(VIRGINIA-120 tpd)		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	PB, Ni Cr, Be	
	Controlled	30 yr.		Pb, Cr Hg, Ni	Be		Pb, Cr Hg, Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni		Local waters at Virginia are tidal (not potable)			Hg	Pb, Ni Cr, Be	
Projected	Baseline	30 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
(WESTERN FLORIDA-3,000 tpd)		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
	Controlled	30 yr.	Hg	Pb, Ni Cr	Be	Hg	Pb, Ni Cr, Be			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	
		100 yr.	Hg	Pb, Ni Cr	Be	Pb Hg	Cr, Be Ni			Pb, Cr Hg, Ni Be		Hg	Pb, Ni Cr, Be	

Be = Beryllium Cr = Chromium Hg = Mercury Ni = Nickel Pb = Lead

TABLE 3-19. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL HUMAN HEALTH EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS
(Human Risk by Ingestion - Organic Compounds)

Facility	Control Scenario	Deposition Interval	Food Ingestion			Soil Ingestion			Surface Water Ingestion			Fish Ingestion		
			Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
(VIRGINIA-120 tpd)		100 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
	Controlled	30 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
		100 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		Local waters at Virginia are tidal (not potable)			PCDD CB PCB	FA	
Projected	Baseline	30 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD CB PCB	FA	
(WESTERN FLORIDA-3,000 tpd)		100 yr.	PCDD CB PCB	FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD CB PCB	FA	
	Controlled	30 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA	
		100 yr.	PCDD CB	PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA		PCDD	CB PCB FA	

CB = Chlorobenzenes   FA = Formaldehyde   PCB = Polychlorinated benzenes   PCDD = Polychlorinated dibenzo-p-dioxins deposited on the surface to underground aquifers. Soils containing a greater fraction of humus matter and having a higher carbon content may impede this downward movement of pollutants.

The analysis described the maximally exposed individuals as a rural farm family of two adults and two young children who partially subsisted on foodstuffs gained by domestic agriculture, freshwater fish caught in local waters, and potable water from local aquifers and surface water sources. The farm family lived and undertook all their activities in the area of maximum dry and wet surface deposition of the pollutants emitted from the stacks of the municipal waste combustor, which was approximately 200 meters from the facility. It is extremely likely that this is an atypical situation, although individuals have been observed to reside as close as 200 meters from municipal waste combustors. It is not expected that individuals will gain most of their food supply from a location this close to the facility.

For the food ingestion pathway, the preliminary methodology predicted a low likelihood of significant contribution to total exposure near the existing and projected model facilities for lead, chromium, nickel, and formaldehyde under all deposition and control scenarios. For mercury and PCB, possible significant contributions to total exposure from food ingestion near both model facilities were predicted for some deposition intervals and control scenarios, while for other deposition intervals and control scenarios the likelihood of significant contribution to total exposure was predicted to be low. In the case of mercury emissions, possible significant contributions to total exposure from food ingestion were predicted for the projected western Florida model facility for all deposition intervals and control scenarios. At the existing Virginia model facility, possible significant contribution to total mercury exposure from food ingestion was predicted only for the 100-year deposition interval for both control scenarios. In the case of PCB emissions, possible significant contributions to total exposure near both model facility were predicted only under the baseline scenario. Under the controlled scenario at the two modeled facilities, a low likelihood of significant contribution to total PCB exposure was predicted for the food ingestion pathway. Of the pollutants modeled, only indirect exposure to PCDD and chlorobenzene was predicted to result in a significant contribution to total exposure from food ingestion near both model facilities for all deposition intervals and control scenarios modeled. No data were available to classify potential contribution to total exposure from food ingestion of beryllium.

For the soil ingestion pathway, a low likelihood of significant contribution to total exposure was predicted for all deposition intervals and control scenarios near the existing and projected model facilities for beryllium, chromium, nickel, chlorobenzene, PCB and formaldehyde. For lead and mercury, the predicted likelihood of significant contribution to total exposure from indirect exposure through soil ingestion ranged from possible to low likelihood for the two model facilities depending on the deposition interval and control scenario. In the case of lead, the preliminary methodology predicted a possible significant contribution to total exposure from soil ingestion for all deposition intervals and control scenarios with the following exception. For both model facilities, analysis of the 30-year deposition interval under the controlled scenario resulted in a prediction of low likelihood of significant contribution to total exposure. The analysis predicted similar results for mercury emissions, except that, unlike for lead, a possible significant contribution was predicted for the 30-year deposition interval under the controlled scenario at the projected western Florida model facility. For the soil ingestion pathway, only indirect exposure to PCDD was predicted to result in a significant contribution to total exposure near the existing and projected model facilities for all deposition intervals and control scenarios.

The potential for indirect exposure to municipal waste combustor emissions from surface runoff and groundwater infiltration was not evaluated for the Virginia facility. This facility is located in a coastal region where runoff of contaminated soil would affect only nonpotable, tidal water bodies. For the projected model facility in Western Florida, the preliminary methodology predicted a low likelihood of significant contribution to total exposure from surface water ingestion for lead, chromium, mercury, nickel, beryllium, chlorobenzene, PCB, and formaldehyde. A possible significant contribution was predicted for PCDD under all scenarios.

For the fish ingestion pathway, a low likelihood of significant contribution to total exposure near the existing and projected model facilities was predicted for lead, nickel, beryllium, chromium and formaldehyde for all deposition intervals and control scenarios. A possible significant contribution to total exposure from fish ingestion near both model facilities was predicted for mercury, PCDD, chlorobenzenes, and PCB for all deposition intervals and control scenarios with the following exception. The analysis predicted that for the projected model facility in western Florida, possible significant contribution to total exposure from fish ingestion for chlorobenzenes and PCB would occur only under the baseline scenario.

An example of the effect of the deposition interval on the likelihood that a particular indirect exposure pathway may contribute significantly to total exposure may be seen in the case of mercury emissions. Indirect exposure through the food ingestion pathway at the existing Virginia facility is predicted to result in possible significant contribution to total exposure for the 100-year deposition interval, but not for the 30-year deposition interval, for both control scenarios. However, this was not observed for the western Florida model facility. In the majority of pollutants analyzed, the length of deposition interval did not affect the predicted likelihood that a particular indirect exposure pathway would contribute significantly to total exposure.

The effect of control scenario on the predicted likelihood that an indirect exposure pathway would contribute significantly to overall exposure can also be seen in this limited analysis. For example, indirect exposure to PCB from food ingestion near both model facilities represents a possible significant contribution to total exposure under the baseline scenario (electrostatic precipitators only) but is reduced to a low likelihood under the controlled scenario (dry alkaline scrubbers combined with fabric filters). Similarly, for the fish ingestion pathway, the likelihood of significant contribution to total exposure posed by emissions of chlorobenzenes and PCB are classified as possible under the baseline scenario but as low likelihood under the controlled scenario.

A similar analysis was performed using the preliminary indirect exposure methodology to predict the likelihood of potential ecological effects from long-term deposition of municipal waste combustor emissions. The likelihood of adverse impacts to terrestrial and aquatic life were analyzed for the Virginia and western Florida model facilities for 30- and 100-year deposition intervals and for baseline and controlled scenarios.

Table 3-20 summarizes the analysis of potential ecological effects near the modeled facilities. Based on the preliminary indirect exposure methodology, possible terrestrial and aquatic impacts were predicted for emissions of lead and mercury for some, but not all, of the deposition intervals and control scenarios modeled for both facilities. Possible terrestrial impacts from lead emissions were predicted for the 100-year deposition interval under the baseline scenario for the existing Virginia facility and for all scenarios except the 30-year controlled scenario for the projected western Florida model facility. In the case of mercury emissions, possible terrestrial impacts were predicted for the 100-year deposition interval under both the baseline and controlled scenarios for the existing Virginia facility, and for all scenarios for the projected western Florida facility. Insufficient data were available to apply the preliminary methodology to predict the likelihood of terrestrial impacts from beryllium. Data to analyze likelihood of terrestrial impacts from chromium were available for certain terrestrial pathways but not for others. Where data were available, a low likelihood of terrestrial impacts was predicted for chromium. A low likelihood of terrestrial impacts was also predicted for nickel under all scenarios.

Possible aquatic effects from lead and mercury emissions were predicted for all deposition intervals and control scenarios for the existing Virginia facility. For the projected western Florida model facility, however, possible aquatic effects were predicted for all scenarios for mercury, while a low likelihood of aquatic effects were predicted under all scenarios for lead. A low likelihood of aquatic impacts was predicted under all scenarios for the two model facilities for beryllium and nickel. As with the analysis of terrestrial impacts, data on chromium were available for some aquatic pathways but not for others. For those pathways analyzed for chromium, a low likelihood of aquatic impacts was predicted.

TABLE 3-20. RESULTS OF ANALYSIS OF THE LIKELIHOOD OF POTENTIAL ECOLOGICAL EFFECTS FROM INDIRECT EXPOSURE TO MUNICIPAL WASTE COMBUSTOR EMISSIONS
(Inorganic Compounds)

			Ecological Risks					
			Terrestrial Impacts			Aquatic Impacts		
Facility	Control Scenario	Deposition Interval	Possible	Low Likelihood	No Data	Possible	Low Likelihood	No Data
Existing	Baseline	30 yr.		Pb, Cr Hg, Ni	Cr* Be	Pb Hg	Cr, Be Ni	Cr*
(VIRGINIA-120 tpd)		100 yr.	Pb Hg	Cr Ni	Cr Be	Pb Hg	Cr, Be Ni	Cr
	Controlled	30 yr.		Pb, Ni Cr	Cr Be	Pb Hg	Cr, Be Ni	Cr
		100 yr.	Hg	Pb, Ni Cr	Cr Be	Pb Hg	Cr, Be Ni	Cr
Projected	Baseline	30 yr.	Pb Hg	Pb, Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr
(WESTERN FLORIDA-3000 tpd)		100 yr.	Pb Hg	Cr, Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr
	Controlled	30 yr.	Hg	Pb, Ni Cr	Cr Be	Hg	Pb, Ni Cr, Be	Cr
		100 yr.	Hg Pb	Cr Ni	Cr Be	Hg	Pb, Ni Cr, Be	Cr

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[*] Data on chromium were available for certain terrestrial and aquatic pathways showing low likelihood of effects, but were not available for other pathways (e.g. toxicity thresholds for soil biota and for predators of aquatic organisms).

Be = Beryllium Cr = Chromium Hg = Mercury Ni = Nickel Pb = Lead

The limited and qualitative aspects of the indirect exposure analysis has generally demonstrated that exposure via these pathways can be comparable in magnitude to exposure to pollutants through the direct inhalation of ambient air. Thus the analysis, although not quantitative, has served the purpose of highlighting the importance of considering all possible routes that humans may be exposed to pollutant emissions from combustion sources. In this regard, the EPA is continuing to develop and improve the current methodology so that routine and reliable estimates of exposure and risk can be made.

3.4 REFERENCES FOR SECTION 3.0

1. Midwest Research Institute. Municipal Waste Combustion Study: Data Base for Municipal Waste Combustors. Review Draft. EPA Contract No. 68-02-3817, ESED Project No. 86/19. January 7, 1987.

2. Reference 1.

3. U. S. Environmental Protection Agency. Municipal Waste Combustion Multipollutant Study. EMB Report No. 86-MIN-02. April 1987.

4. A. J. Fossa, R. S. Kerr, A. S. Columbus, and R. Waterfall. Air Emissions Characterization of Municipal Waste Combustors in New York State. New York State Department of Environmental Conservation. Presented at the 80th Annual Meeting of Air Pollution Control Association. June 21-26, 1987. New York, New York.

5. Reference 1.

6. Klicius, R., D. J. Hay, and A. Finkelstein. The National Incineration Testing and Evaluation Program. Presented for use by EPA's Science Advisory Board by Environment Canada. Ontario, Canada. September 1986.

7. Reference 6.

8. Determination of ChloroDibenzo-p-dioxins, Chlorinated Dibenzofurans, Chlorinated Biphenyls, Chlorobenzenes and Chlorophenols in Air Emissions at SWARU. Report No. ARB-02-84-ETRD. Ministry of Environment. Ontario, Canada. 1984.

9. U. S. Environmental Protection Agency. Environmental Assessment of Waste-to-Energy Process: Braintree Municipal Incinerator. EPA 600/7-80-149. 1980.

10. Hahn, J. L. Air Emissions Testing at the Wurzburg, West Germany Waste-to-Energy Facility. Presented at the Annual Meeting of the Air Pollution Control Association, June 1986.

11. Radian Corporation. Municipal Waste Combustion Study Data Gathering Phase. Preliminary Draft. EPA Contract No. 68-02-3818. 1985.

12. Hart, F. C., and Associates. Assessment of Potential Public Health Impacts Associated with Predicted Emissions of Polychlorinated Dibenzodioxins and Polychlorinated Dibenzofurans from the Brooklyn Navy Yard Resource Recovery Facility. Prepared for the New York City Department of Sanitation. 1984.

13. U. S. Environmental Protection Agency. Municipal Waste Combustion Study: Characterization of the Municipal Waste Combustion Industry. EPA/530-SW-87-021h. March 1987.

14. 51 FR 33992. Guidelines for Carcinogen Risk Assessment. September 24, 1986.

15. U. S. Environmental Protection Agency. Mercury Health Effects Update. Final Report. EPA 600/8-84-019F. August 1984.

16. American Conference of Government Industrial Hygienists. Threshold Limit Values and Biological Exposure Indices for 1986-1987. ISBN:0-936712-69-4. Cincinnati, Ohio.

17. Letter from Henry, J. (Texas Air Control Board), to Kellam, R. (U.S. Environmental Protection Agency). June 26, 1987. Information on ambient levels of HCl associated with corrosion.

APPENDIX A
HUMAN EXPOSURE MODEL

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HUMAN EXPOSURE MODEL

A.1 INTRODUCTION

The Human Exposure Model (HEM) is a general model designed to estimate the population exposed to air pollutants emitted from stationary sources and the carcinogenic risk associated with this exposure. The HEM is comprised of (1) an atmospheric dispersion model, with included meteorological data, (2) a population distribution based on Bureau of Census data, and (3) a procedure for estimating risks due to the predicted exposure. The only inputs needed to operate this model are source data, e.g., plant location, height of the emission release point, the temperature of the off gases. Based on source data, the model estimates the magnitude and distribution of ambient air concentrations of the pollutant in the vicinity of the source. These concentration estimates are coupled with the population to estimate public exposure to the pollutant. The HEM then predicts population risks if a unit risk number determined from health data is input for the pollutant.

Within HEM there are two basic models, the Systems Applications Human Exposure and Dosage (SHED) model and the Systems Application Human Exposure and Risk (SHEAR) model. The SHED model is used primarily for major point sources where the latitude and longitude are known. It is commonly used to evaluate exposure and risk for a source category on a nationwide basis. In contrast the SHEAR model is used for regional analysis of multiple point sources and area sources. It is also used to model "prototype" sources where it is impractical to specify latitude and longitude for individual sources.

For analysis of nationwide exposure and cancer risks from municipal waste combustor emissions, the SHED model option in the HEM was used. The following data are used by the HEM to assess exposures to and dosages of atmospheric chemicals:

- source location
- emissions data
- vent parameters
- atmospheric reactivities of chemicals
- meteorological data
- population distribution data

In addition to these inputs, health data are needed to derive population risks from population exposure. However, these data are provided in the form of a unit risk estimate, which is derived external to the exposure model. These health data are not presented in this appendix.

The remainder of this appendix describes the input data required to run the SHED option of the HEM and how the model estimates exposure and dosage. A discussion of uncertainties and assumptions inherent in the HEM is also given.

A.2 INPUT DATA

A.2.1 Source Location

The geographic location of the source must be specified in terms of latitude and longitude.

A.2.2 Emissions Data

The emission rate in Mg/yr must be specified for each modeled pollutant. (When using SHED, separate modeling runs are required to model exposure to different pollutants.)

A.2.3 Vent Parameters

Vent parameter data are necessary for dispersion modeling of emissions. Vent parameter data required include the number of vents, vent height, vent diameter, gas discharge temperature, gas emission velocity, fugitive discharge area (if applicable), and building cross-sectional area.

A.2.4 Atmospheric Reactivities of Chemicals

The HEM is generally constrained to a study area in a 50 km radius of the source. In most instances, the reactivity of a chemical is not sufficiently high to cause significant removal before the material is dispersed to this distance. For these chemicals. this input to the HEM can be left blank with the model assuming no atmospheric decay. Some chemicals (e.g., formaldehyde) do have high reactivity and should be analyzed for this behavior. When applicable, atmospheric reactivity is specified in terms of daytime and nighttime decay rate (min^{-1}).

A.2.5 Meteorological Data

The dispersion computations performed by the HEM require data on wind speed, wind direction, and the intensity of atmospheric turbulence. The turbulence intensity is represented by the atmospheric stability class. These data are provided by a permanent data base in the HEM known as the Stability Array or STAR data set.

Martin and Tikvart developed the STAR program from routinely collected meteorological data to generate frequencies and percentage frequencies of wind direction by speed classes for each stability category.[1] The specifications of stability categories depending on wind speed and sky cover were set up by Pasquill and were modified by Turner.[2,3] The program was adopted for use at the National Climatic Center (NCC), where archived records of all national reporting weather stations are kept. The most up-to-date version of the STAR data from all STAR stations in the country were used to produce the matrices of STAR frequencies used in the HEM. The format of the STAR data includes sixteen wind directions, six wind speed classes, and seven stability categories with categories A, B, C, and D_{day} in the daytime and categories D_{night}, E, and F in the nighttime. There are data sets for 312 stations in the STAR data file. Figure A-1 shows the locations of these STAR sites.

By default, meteorological data recorded at the STAR station nearest to the source are used in the dispersion modeling for SHED. However, local meteorological trends and topographic features may be more important factors

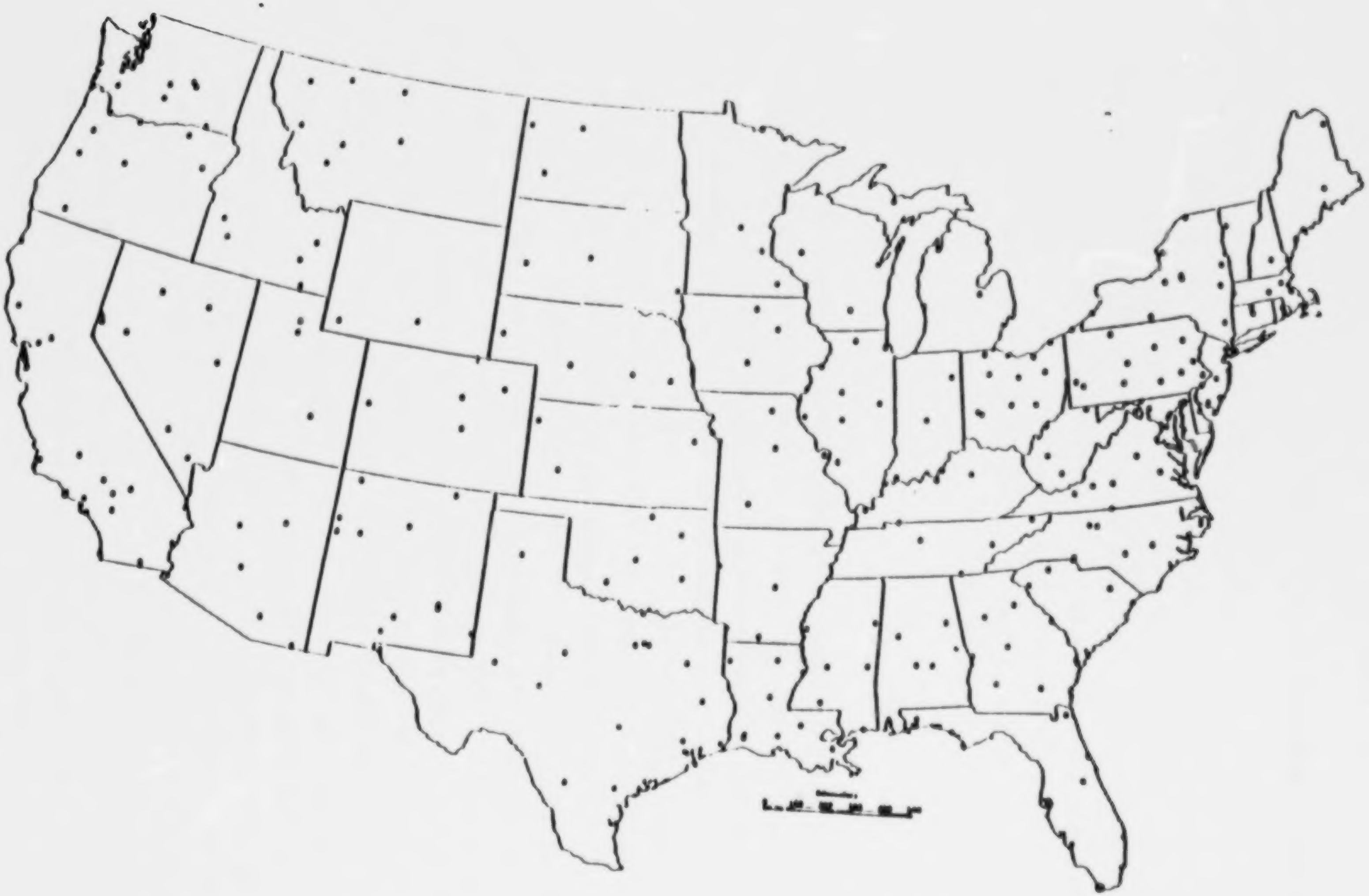


Figure A-1. Location of STAR Sites.

in selecting a STAR station than is the absolute distance between the source and the station. A STAR station with climatologial conditions most similar to those of the source of emissions may not be the nearest station, so the STAR stations may be manually selected to account for such situations.

A.2.6 Population Data

The HEM program uses the latitude and longitude from the input data in determining the population of the study area. The permanent data base is comprised of the 1980 Census Data Base broken down by Block Group/ Enumeration District (BG/ED). The population data base contains the population centroid coordinates (latitude and longitude) and the 1980 population of each BG/ED in the country (about 300,000 centroids in 50 states plus the District of Columbia).

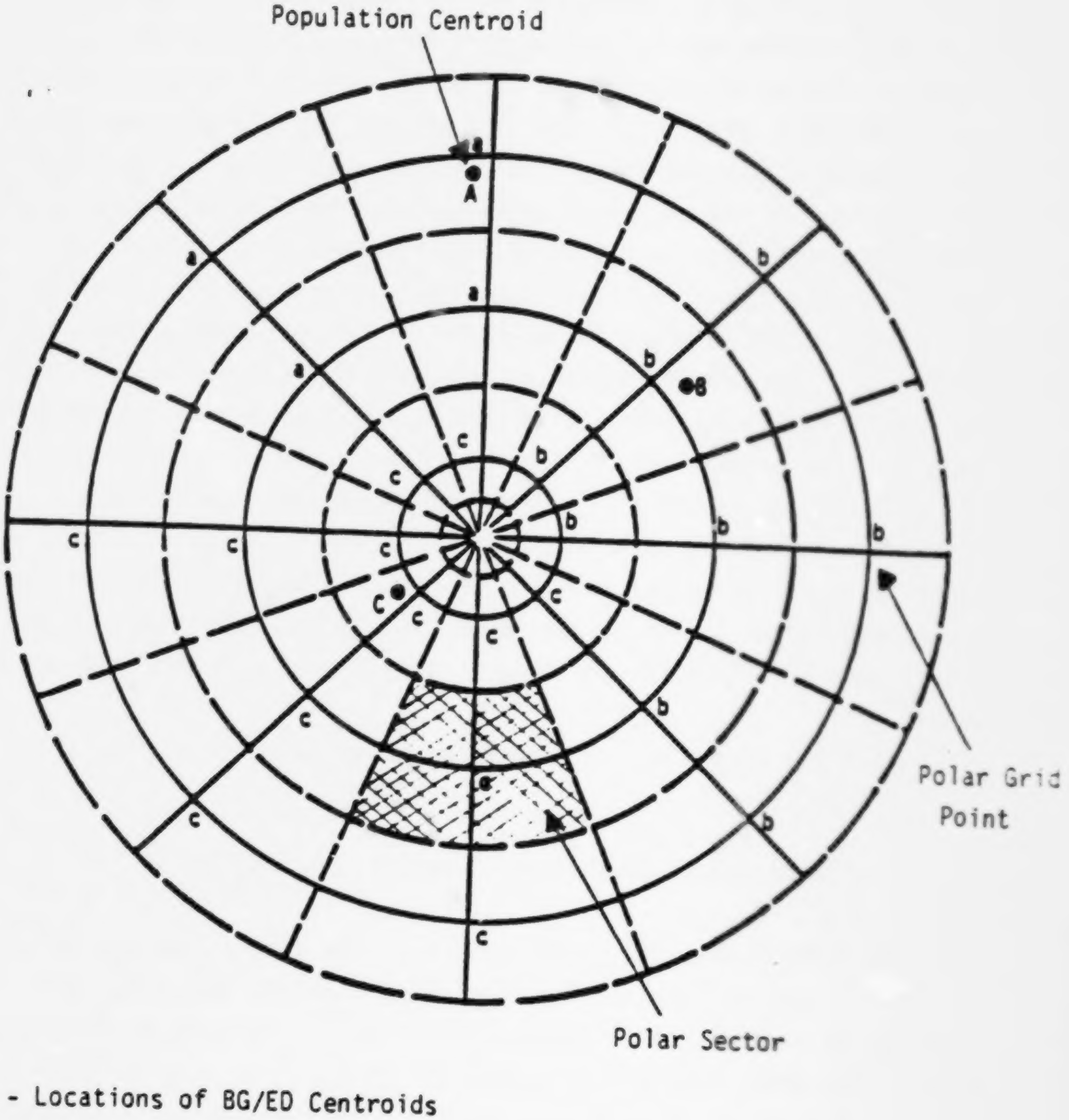
A.3 EXPOSURE AND DOSAGE ESTIMATE SCHEME

The output of the HEM is a concentration array for 160 receptors around the plant (10 receptors along each of the 16 wind directions). These are the sum of concentration patterns resulting from all sources within a plant. The basic approach used in combining the concentration pattern with the population distribution pattern around a plant is explained in this subsection. Three terms are defined here that will be used frequently in the following discussion. A polar grid point is one of the 160 receptors at which concentrations are estimated by the dispersion modeling. A population centroid is the population-weighted geographical center of a BG/ED for which geodetic coordinates are known. A grid cell is defined as the area bounded by two radial arcs and two wind directions.

Exposure is the product of the population and the concentration to which that population is exposed. To form this product, both the concentration and the population must be known at the same location or point. The SHED model uses a two-level interpolation scheme to pair the concentrations with populations prior to the computation of dosages and exposures. The two-level approach is appropriate because the concentrations are defined on a radius-azimuth (polar) grid pattern with non-uniform spacing. This means that at small radii the grid cells are much smaller than BG/EDs while at larger radii the grid cells are much larger than BG/EDs. Interpolation techniques are most appropriately applied by interpolating values of the factor defined on a coarse network (larger) at the locations of the finer (smaller) network, thus maximizing the resolution and minimizing the uncertainties of interpolation. As previously mentioned, the fine/coarse relationship between polar grid cells and BG/EDs varies with radius. Hence, the two-level approach allows the BG/ED population to be interpolated to the grid point when the BG/EDs are larger than the grid cells and allows the grid point concentration to be interpolated to the BG/ED centroid when the reverse is true. The details of this approach are outlined below. The numbers used assume the analysis is to be completed for a 50 km radius. If the maximum radius changes, so will the radii of the grid points.

For BG/ED centroids located between 0.2 km and 3.5 km from the source, populations are apportioned among neighboring polar grid points. There are 64 (4 x 16) polar grid points within this range. Associated with each of these grid points, at which the concentration is known, is a smaller polar sector bounded by two concentric arcs and two radial lines. The boundary concentric arcs are defined by radii of .10, .35, .75, 1.5, and 3.5 km and the boundary radial lines are drawn right in the middle of two wind directions. These boundary lines are represented by the dashed lines in Figure A-2. Each of the polar grid points are assigned to the nearest BG/ED centroid identified from the census data set. The population at each centroid is then apportioned among all polar grid points assigned to that centroid according to the area of the polar sector associated with the grid point. For example, all of the polar grid points (labeled b) nearest population centroid B would be assigned a certain proportion of the people in B. The ratio between the area of the polar sector and the area assigned to the population centroid determine the proportionment. Thus, the population density is assumed to be the same for all polar sectors assigned to a single centroid. Figure A-2 shows that the grid points closer to the source are boxed in by much smaller sectors than those further away. Hence

Figure A-2. Pairing of BG/ED with Concentration Within A 3.5 Km Radius



Population Centroid
A
B
C
a
b
c
Polar Grid Point
Polar Sector

A, B, C - Locations of BG/ED Centroids

a, b, c - Polar grid points to be allocated a portion of population reported at centroids A, B, C, respectively.

the b grid points closer to the center will receive a smaller apportionment of the people from population centroid B than those further away. Both concentration and population counts are thus available for each polar grid point.

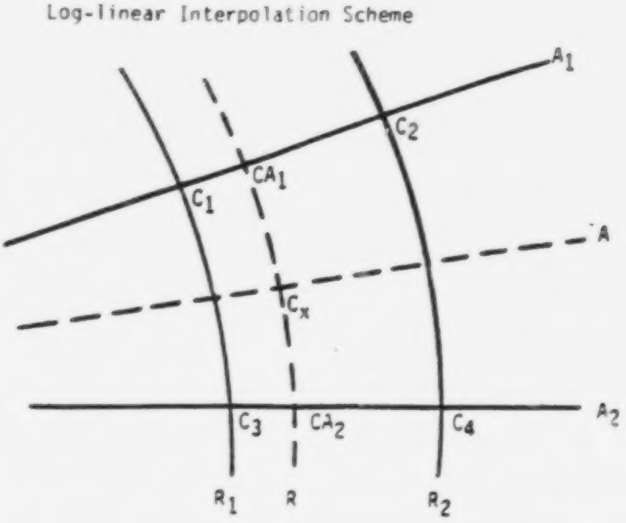
Log-log linear interpolation is used to estimate the concentration at each BG/ED population centroid located between 3.5 km and 50 km from the source. Concentration estimates for 96 (6 x 16) grid points (receptors at 5.0, 10.0, 20.0, 30.0, 40.0, and 50.0 km from the source along each of the sixteen directions) resulting from dispersion modeling are used here as reference points for this interpolation. For each BG/ED centroid, four reference points are located at the four corners of the polar sector in which the centroid is located. These four reference points (labeled C_1, C_2, C_3, and C_4) would surround the centroid (C_x) as depicted in Figure A-3. There is a linear relationship between the logarithm of concentrations and the logarithm of distances for receptors more than 2.0 km away from the source. This relationship is used to estimate the concentrations at points CA_1 and CA_2 (see Figure A-3). These estimates are then linearly interpolated with the polar angle to determine the concentration at the centroid (C_x). Using the two-level approach, concentrations and populations are paired up for the 64 concentration grid points within 3.5 km of the source and for all BG/ED centroids located between 3.5 km and 50 km from the source.

The total dosage was then computed as follows:

$$\text{Total Annual Dosage (ug/m}^3\text{-person)} = \sum_{i=1}^{N} P_i Ci \quad ,$$

where P_i = the population at point i, C_i = the annual average concentration at point i, and N = the total number of grid points and BG/ED centroids with a specified combination of concentration and population (representing the entire area within 50 km of the source).

The population exposed to each of a number of concentration levels, L_j, was computed by:



Given:

A = The angle in degrees subtended clockwise about the source from due south to the BG/ED centroid;

A_1 = The angle from due south to the radial line immediately counterclockwise from A, or passing through A if there is an exact match;

A_2 = The angle from the south to the radial line immediately clockwise of A_1 (A_2 is 0 if it is due south);

R = The distance from the source to the BG/ED centroid;

R_1 = The distance from the source to the largest circular arc of radius on grid that is less than R;

R_2 = The distance from the source to the smallest circular arc of radius on grid that is greater than or equal to R;

C_1 = The concentration value at (A_1, R_1);

C_2 = The concentration value at (A_1, R_2);

C_3 = The concentration value at (A_2, R_1);

C_4 = The concentration value at (A_2, R_2); then

$$CA_1 = \exp (\ln C_1 + (\ln C_2 - \ln C_1) (\ln R - \ln R_1)/(\ln R_2 - \ln R_1)$$

$$CA_2 = \exp (\ln C_3 + (\ln C_4 - \ln C_3) (\ln R - \ln R_1)/(\ln R_2 - \ln R_1)$$

$$C_x = CA_1 + (CA_2 - CA_1)(A - A_1)/(A_2 - A_1)$$

Figure A-3. Log-linear Interpolation Scheme

$$\text{Exposure to } L_j \text{ (person)} = \sum_{i=1}^{N} P_i S_i(C_i, L_j) \quad ,$$

where

$$S_i = \begin{cases} 0 \quad , & \text{if } C_1 < L_j \\ 1 \quad , & \text{if } C^i \geq L_j \end{cases} \quad .$$

The dosage of the fraction of the population that is exposed to concentrations greater than or equal to each of a number of concentration levels, L_j, was computed by using the following summation:

$$\text{Annual Dosage at } L_j \text{ (}\mu\text{g/m}^3\text{-person)} = \sum_{i=1}^{N} P_i C_i S_i(C_i, L_j) \quad .$$

Note that the annual dosage at the minimum concentration within 50 km of the source will equal the total annual dosage.

The concentration levels at which exposure is to be estimated are selected by an exponential function coded in the program.

A.4 UNCERTAINTIES AND ASSUMPTIONS

The HEM estimates concentrations of a substance in the ambient air at specific points around a source. It is known, however, that pollutant exposure occurs for exposures other than ambient air exposures (i.e., food, water) and populations move through different microenvironments. Accordingly, the HEM employs a number of simplifying assumptions including:

1. It is assumed that most exposure occurs at population-weighted centers (centroids) of block group and enumeration districts (BG/ED) as the locations of actual residences are not contained in available databases. The model relies on information provided in a database developed by the U.S. Census Bureau.
2. It is assumed that people reside at these centroids for their entire lifetimes (assumed to be 70 years for calculating cancer risk).

3. It is assumed that indoor concentrations are the same as outdoor concentrations.

4. It is assumed that plants emit pollutants at the same emission rate for 70 years.

5. It is assumed that the only source of exposure is the ambient air and resuspension of pollutants via dust is not considered.

6. It is assumed that there is no population migration or growth.

7. The model does not provide for discriminating exposure situations that may differ with age, sex, health status, or other situations.

8. The model is designed to model for flat terrain.

There is also uncertainty associated with various specifications required for running the HEM as well as uncertainty associated with the variables specified. These model specifications and variables include:

1. Given the variations in meteorology each year and with each location, there is uncertainty as to the extent to which the meteorological station used to model the site modeled is representative. This includes the selection of urban or rural for stability.

2. There is uncertainty associated with the emission estimates and the plant parameters used to characterize the emission source.

3. If the plant is not correctly located, there will be a problem in the matching of population census data with concentrations, which alters risk estimates.

Simplifying assumptions are made by the HEM in locating population and computing exposure.

In the case of locating population, the SHED model uses census data to the BG/ED level. Although this is the smallest defined population unit, it still places on average approximately 800 people at one point. In reality, some people will reside closer to the plant and may be more highly exposed. For others, the modeled exposure will be an overestimate of the actual exposure.

Another problem occurs when two plants are located within 50 km of one another. The people who are exposed to both plants will be counted twice in the summation of the exposed population. Although the number of people will be an overestimate, the dosage (people times concentrations) is not affected by the overlap of the study areas. Two people each exposed to a concentration x from one of two plants is regarded as the same amount of exposure as a single person exposed to a concentration of x from both. Populations may be partially doublecounted if sources are close to each other, but aggregate exposure is not.

The SHED model may also mislocate people if a plant is located near a large body of water. The manner in which SHED spreads the population and concentrations within the 3.5 km radius allows the program to assign people to areas which may be covered by bodies of water. One potential problem may arise if the program determines a maximum concentration to which people are exposed that is located over a lake or other body of water. In this case, it could be argued that no one is actually exposed to this concentration. For those sources that appear to produce maximum concentrations, identification of the receptor location and consultation of topographical maps to verify public exposure is recommended. If the maximum receptor is located in an area where people would not reside, the next highest receptor should be used or that plant should be modeled individually with a different, more specific model.

The concentration patterns used in the exposure computations are obtained through atmospheric dispersion modeling based on known source characteristics and weather patterns at nearby stations. Naturally, any deviations in these estimates from the true pattern directly affect the exposure results. Thus, if the nearest STAR station is 50 to 100 miles away, the weather patterns may not be representative of the weather patterns in the area around the source.

The program also assumes flat terrain. This is especially crucial when dealing with sources in a valley/mountainous area because the STAR station will not give adequate information to model the wind patterns accurately.

In addition, the SHED model uses a Gaussian dispersion model to determine the concentrations around the source. This model is based on how gases will act when dispersed from a stack. Thus, if the pollutant is fibrous or particulate, such as asbestos, the assumption that the pollutant behaves as a gas may misrepresent the actual dispersion pattern.

The assumption is also made that the annual average weather conditions continually persist and that the source emits at the annual average amount. At times, these averages may not be representative over shorter terms. It also assumes that each recorded wind persists in a given direction long enough to yield dispersion out to 50 km.

There are also time-dependent aspects of the exposure problem. The exposure program uses a time-averaged concentration pattern for each source, so that the time dimension is ignored in the computations. If the population distribution were essentially constant over the averaging time period, the resulting estimates would be true averages. However, population distributions are constantly changing as people commute to work, go shopping, and take longer trips. Particularly in urban industrial centers, the shifts in populations and concentrations throughout the day may be highly correlated; thus, the actual exposure may differ considerably from the value obtained by matching time-averaged concentrations with population distributions based on census addresses. Whether the exposure is over- or under-estimated depends on whether populations in the vicinity of a source are drained (e.g., because people leave residences near the source for work in an urban center) or are augmented (e.g., because of employment near the source).

A.5 REFERENCES FOR APPENDIX A

1. Martin, D. O., and J. A. Tikvart. A General Atmospheric Diffusion Model for Estimating the Effects on Air Quality of One or More Sources. Presented at the 61st Annual Air Pollution Control Association Meeting. St. Paul, Minnesota. June 1968.

2. Pasquill, F. The Estimation of the Dispersion of Windborne Material. Meteorology 90:33-49. 1961.

3. Turner, D. B. A Diffusion Model for an Urban Area, Atmospheric Dispersion Estimates. 6th Printing Rev., Government Printing Office, U. S. Environmental Protection Agency, Publication No. AP-26. 1964.

APPENDIX B

INDUSTRIAL SOURCE COMPLEX SHORT-TERM AIR DISPERSION MODEL

APPENDIX B
INDUSTRIAL SOURCE COMPLEX SHORT-TERM AIR DISPERSION MODEL

B.1 INTRODUCTION

The Industrial Source Complex (ISC) Dispersion Model combines and enhances various dispersion model algorithms into a set of two computer programs that can be used to assess the air quality impact of emissions from the wide variety of sources associated with an industrial source complex. For plumes comprised of particulates with appreciable gravitational settling velocities, the ISC Model accounts for the effects on ambient particulate concentrations of gravitational settling and dry deposition. Alternatively, the ISC Model can be used to calculate dry deposition. The ISC Short-Term Model (ISCST), an extended version of the Single Source (CRSTER) Model, is designed to calculate concentration or deposition values for time periods of 1, 2, 3, 4, 6, 8, 12, and 24 hours.[1] If used with a year of sequential hourly meteorological data, ISCST can also calculate annual concentration or deposition values. The ISC Long-Term Model (ISCLT) is a sector-averaged model that extends and combines basic features of the Air Quality Display Model (AQDM) and the Climatological Dispersion Model (CDM). The Long-Term Model uses statistical wind summaries to calculate seasonal (quarterly) and/or annual ground-level concentration or deposition values. Both ISCST and ISCLT use either a polar or a Cartesian receptor grid. The major features of the ISC Model are listed in Table B-1.

The ISC Model programs accept the following source types: stack, area and volume. The volume source option is also used to simulate line sources. The steady-state Gaussian plume equation for a continuous source is used to calculate ground-level concentrations for stack and volume sources. The area source equation in the ISCST Model programs is based on the equation for a continuous and finite cross-wind line source. In the ISCLT Model program, the area source treatment uses a virtual point source approximation. The generalized Briggs plume-rise formulas are used to

TABLE B-1. MAJOR FEATURES OF THE ISC MODEL

Polar or Cartesian coordinate systems.

Rural or one of three urban options.

Plume rise due to momentum and buoyancy as a function of downwind distance for stack emissions.[a]

Procedures for evaluating building wake effects.[b]

Procedures for evaluating stack-tip downwash.[c]

Separation of multiple point sources.

Consideration of the effects of gravitational settling and dry deposition on ambient particulate concentrations.

Capability of simulating point, line, volume and area sources.

Capability to calculate dry deposition.

Variation with height of wind speed (wind-profile exponent law).

Concentration estimates for 1-hour to annual average.

Terrain-adjustment procedures for elevated terrain including a terrain truncation algorithm.

Consideration of time-dependent exponential decay of pollutants.

Method to account for buoyancy-induced dispersion.[d]

A regulatory default option to set various model options and parameters to EPA recommended values.

Procedure for calm-wind processing.

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[a] References 2, 3, 4, 5 and 6.

[b] References 7 and 8.

[c] Reference 5.

[d] Reference 9.

calculate final as well as gradual plume rise.[10,11,12,13,14] Procedures suggested by Huber and Snyder[15] and Huber[16] are used to evaluate the effects of the aerodynamic wakes and eddies formed by buildings and other structures on plume dispersion. A wind-profile exponent law is used to adjust the observed mean wind speed from the measurement height to the emission height for the plume rise and concentration calculations. Procedures utilized by the Single Source (CRSTER) Model are used to account for variations in terrain height over the receptor grid. Except for Urban Mode 3, the Pasquill-Gifford curves[17] are used to calculate lateral (o_y) and vertical (o_z) plume speed. The ISC Model has one rural and three urban options. In the Rural Mode, rural mixing heights and the o_y and o_z values for the indicated stability category are used in the calculations. In Urban Mode 1, the stable E and F stability categories are redefined as neutral D stability. In Urban Mode 2, the E and F stability categories are combined and the o_y and o_z values for the stability category one step more unstable than the indicated stability category (except A) are used. In Urban Mode 3, the Briggs urban dispersion coefficients derived from McElroy-Pooler observations are used. Urban mixing heights are used in all three urban modes.

For purposes of analyzing deposition of municipal waste combustor emissions, the ISC Short-Term Model was used. The ISCST dry deposition model is based on the Dumbauld et al. deposition model.[18] This model, which is an advanced version of the Cramer et al. deposition model,[19] assumes that a fraction y_n of the material that comes into contact with the ground surface by the combined processes of atmospheric turbulence and gravitational settling is reflected from the surface. The reflection coefficient y_n, which is a function of settling velocity and the ground surface for particulates and of the ground surface for gaseous pollutants, is analogous in purpose to the deposition velocity used in other deposition models. The Cramer deposition model has closely matched ground-level deposition patterns for droplets with diameters above about 30 micrometers, while the more generalized Dumbauld et al. deposition model has closely matched observed deposition patterns for both large and small droplets.

The input requirements for the ISCST consist of four categories:

- Meteorological data
- Source data
- Receptor data
- Program control parameters

The remainder of this appendix describes the input data required by the ISCST Model and the manner in which the model predicts dry deposition values. The uncertainties and assumptions in the ISCST Model are also discussed.

B.2 INPUT DATA

B.2.1 Meteorological Data

Table B-2 gives the hourly meteorological inputs required by the ISC Model short-term computer program (ISCST). These inputs include the mean wind speed measured at height z_1, the direction toward which the wind is blowing, the wind-profile exponent, the ambient air temperature, the Pasquill stability category, the vertical potential temperature gradient and the mixing layer height. In general, these inputs are developed from concurrent surface and upper-air meteorological data by the RAMMET preprocessor program as used by the Single Source (CRSTER) Model.[20,21] If the preprocessed meteorological data are used, the user may input, for each combination of wind-speed and Pasquill stability categories, site-specific values of the wind-profile exponent and the vertical potential temperature gradient. If the user does not input site-specific wind-profile exponents and vertical potential temperature gradients, the ISC Model uses the default values given in Table B-3.

The ISCST program has a rural and three urban options. IN the Rural Mode, rural mixing heights and the Pasquill-Gifford (P-G) σ_y and σ_z values for the indicated stability category are used in the calculations. Urban mixing heights are used in the urban modes. In Urban Mode 1, the stable E and F categories are redefined as neutral (D) stability, and the P-G σ_y and

TABLE B-2. HOURLY METEOROLOGICAL INPUTS REQUIRED BY THE ISC SHORT-TERM MODEL PROGRAM

Parameter	Definition
$\bar{u}_1$	Mean wind speed in meters per second (m/sec) at height z_1 (default value for z_1 is 10 meters)
AFVR	Average random flow vector (direction toward which the wind is blowing)
p	Wind-profile exponent (default values assigned on the basis of stability; see Table B-3)
T_a	Ambient air temperature in degrees Kelvin (°K)
H_m	Depth of surface mixing layer (meters), developed from twice-daily mixing height estimates by the meteorological preprocessor program
Stability	Pasquill stability category (1 = A, 2 = B, etc.)
$\frac{a\theta}{az}$	Vertical potential temperature gradient in degrees Kelvin per meter (default values assigned on the basis of stability category; see Table B-3).

TABLE B-3. DEFAULT VALUES FOR THE WIND-PROFILE EXPONENTS AND VERTICAL POTENTIAL TEMPERATURE GRADIENTS

Pasquill Stability Category	Urban Wind-Profile Exponent p	Rural Wind-Profile Exponent p	Vertical Potential Temperature Gradient (^{o}K/m)
A	0.15	0.07	0.000
B	0.15	0.07	0.000
C	0.20	0.10	0.000
D	0.25	0.15	0.000
E	0.30	0.35	0.020
F	0.30	0.55	0.035

σ_z values are used. In Urban Mode 2, the E and F stability categories are combined and the P-G σ_y and σ_z values for the stability category one step more unstable than the indicated category are used in the calculations. For example, the P-G σ_y and σ_z values for C stability are used in calculations for D stability in Urban Mode 2. In Urban Mode 3, stability categories are not combined, but urban dispersion curves of Briggs are used. These curves, as reported by Gifford,[22] were derived from the St. Louis Dispersion Study.[23] Table B-4 gives the dispersion coefficients used in each mode.

The Rural Mode is usually selected for industrial source complexes located in rural areas. However, the urban options may also be considered in modeling an industrial source complex located in a rural area if the source complex is large and contains numerous tall buildings and/or large heat sources (for example, coke ovens). An urban model is appropriate for these cases in order to account for the enhanced turbulence generated during stable meteorological conditions by the surface roughness elements and/or heat sources. If an urban mode is appropriate, Urban Mode 3 is recommended by EPA for regulatory applications. Modes 1 and 2 are generally not used but are available to the user for historical interest and model evaluation.

B.2.2 Source Input Data

Table B-5 summarizes the source input data requirements of the ISCST Model for point sources. Source evaluations above mean sea level and source locations with respect to a user-specified origin are required for all sources. If the Universal Transverse Mercator (UTM) coordinate system is used to define receptor locations, UTM coordinates can only be used to define source locations if a Cartesian receptor array is used. With a polar receptor array, the origin is at (X=0, Y=0). The X and Y coordinates of the other sources with respect to this origin are then obtained from a plant layout drawn to scale. The x axis is positive to the east and the y axis is positive to the north. Note that the origin of the polar receptor array is always at X=0, Y=0.

TABLE B-4. PASQUILL STABILITY CATEGORIES USED BY THE ISC MODEL TO SELECT DISPERSION COEFFICIENTS FOR THE RURAL AND URBAN MODES

Actual Pasquill Stability Category[a]	Pasquill Stability Category for the σ_y, σ_z Values Used in ISC Model Calculations			
	Rural Mode	Urban Mode 1	Urban Mode 2	Urban Mode 3[b]
A	A	A	A	A
B	B	B	A	B
C	C	C	B	C
D	D	D	C	D
E	E	D	D	E
F	F	D	D	F

[a] The ISCST program redefines extremely stable G stability as very stable F stability.

[b] The Briggs urban dispersion curves combine A and B into one "very unstable" category, and E and F into one "stable" category.

TABLE B-5. SOURCE INPUTS REQUIRED BY THE ISCST MODEL PROGRAMS FOR POINT SOURCES

Stacks	Definition
Q	Pollutant emission rate for concentration calculations (mass per unit time)
Q_t	Total pollutant emissions during the period t for which deposition is calculated (mass)
Ψ	Pollutant decay coefficient (seconds^{-1})
X, Y	X and Y coordinates of the stack (meters)
Z_s	Elevation of base of stack (meters above mean sea level)
h	Stack height (meters)
v_s	Stack exit velocity (meters per second)
d	Stack inner diameter (meters)
T_s	Stack exit temperature (degrees Kelvin)
ϕ_n	Mass fraction of particulates in the n^{th} settling-velocity category
V_{sn}	Gravitational settling velocity for particulates in the n^{th} settling-velocity category
γ_n	Surface reflection coefficient for particulates in the n^{th} settling-velocity category
h_b	Height of building adjacent to the stack (meters)
W	Width of building adjacent to the stack (meters)
L	Length of building adjacent to the stack (meters)

The pollutant emission rate is also required for each source. If the pollutant is depleted by any mechanism that can be described by time-dependent exponential decay, the user may enter a decay coefficient Ψ. Note that if SO_2 is modeled in the urban mode, and the regulatory default option is chosen, a decay half life of 4 hours is automatically assigned. The parameters ϕ_n, V_{sn}, and y_n are only input if concentration or deposition calculations are being made for particulates with appreciable gravitational settling velocities (diameters greater than about 20 micrometers). Particulate emissions from each source can be divided by the user into a maximum of 20 gravitational settling-velocity categories. Emission rates may be held constant or may be varied as follows:

- By hour of the day
- By season or month
- By hour of the day and season
- By wind-speed and stability categories (applies to fugitive sources of wind-blown dust)

Emission rates used by the long-term model program ISCLT may be annual average rates or may be varied by season or by wind-speed and stability categories.

Additional source inputs required for stacks include the physical stack heights, the stack exit velocity, the stack inner diameter, and the stack exit temperature. For an area source or a volume source, the dimensions of the source and the effective emission height are entered in place of these parameters. If a stack is located on or adjacent to a building and the stack height to building height ratio is less than 2.5, the length (L) and width (W) of the building are required as source inputs in order to include aerodynamic wake effects in the model calculations.

B.2.3 Receptor Data

The ISCST Model computer programs allow the user to select either a Cartesian (X, Y) or a polar (r, θ) receptor grid system. In the Cartesian system, the x-axis is positive to the east of a user-specified origin and the y-axis is positive to the north. In the polar system, r is the radial distance measured from the origin (X=0, Y=0) and the angle Θ (asimuth bearing) is measured clockwise from north. If the industrial source complex is comprised of multiple sources that are not located at the same point, a Cartesian coordinate system is usually more convenient than the polar coordinate system. Additionally, if the Universal Transverse Mercator (UTM) coordinate system is used to define source locations and/or to extract the elevations of receptor points from USGS topographic maps, the UTM system can also be used in the ISC Model calculations. Discrete (arbitrarily placed) receptor points corresponding to the locations of air quality monitors, elevated terrain features, the property boundaries of the industrial source complex or other points of interest can be used with either coordinate system.

B.3 DRY DEPOSITION CALCULATIONS

Deposition for particulates in the n^{th} settling-velocity category or a gaseous pollutant with zero settling velocity V_{sn} and a reflection coefficient y_n is given by:

$$DEP = K\,Q_t\,V_d D\,(1-y_n)\,\phi_n\,(2\pi\sigma_y\sigma_z x)^{-1}\exp[-0.5(y/\sigma_y)^2] \qquad \text{Equation (B-1)}$$

where:

K = a scaling coefficient to convert calculated concentrations to desired units (default value of 1×10^6 for Q in g/sec and concentration in ug/m^3)

Q_t = total amount of pollutant emitted during the time period t for which the deposition calculation is made

V_d = Vertical Term

D = Decay Term

y_n = reflection coefficient for particulates in the n^{th} settling-velocity category

ϕ_n = mass fraction of particulates in the n^{th} settling-velocity category

σ_y, σ_z = standard deviation of later, vertical concentration distribution (m)
X = downwind distance (m)
Y = crosswind distance (m)

A description of the Vertical Term (V_d), Decay Term (D), reflection coefficient (y_n), and dispersion coefficients (σ_y and σ_z) follows.

B.3.1 Decay Term

The Decay Term, which is a simple method of accounting for pollutant removal by physical or chemical processes, is of the form:

$$\begin{array}{lll} D = \exp(-\Psi x/u) & \text{for } \Psi = 0. & \\ \quad \text{or} & & \text{Equation (B-2)} \\ \quad = 0. & \text{for } \Psi = 0. & \end{array}$$

(i.e., decay not considered when zero is input for Ψ).

where:

Ψ = the decay coefficient (sec^{-1})
x = downwind distance (meters)
u = wind speed (m/s)

For example, if $T_{1/2}$ is the pollutant half life in seconds, Ψ is obtained from the relationship:

$$\Psi = 0.693/T_{1/2} \qquad \text{Equation (B-3)}$$

B.3.2 Dispersion Coefficients

Equations that approximately fit the Pasquill-Gifford curves[24] are used to calculate σ_y (meters) and σ_z (meters) for Urban Modes 1 and 2 and the rural mode. The equations used to calculate σ_y are of the form:

$$\sigma_y = 465.11628 \, (x) \tan(TH) \qquad \text{Equation (B-4)}$$

where:

$$TH = 0.017453293 \, (c - d \ln x) \qquad \text{Equation (B-5)}$$

In Equations (B-4) and (B-5) the downwind distance x is in kilometers, and the coefficients c and d are listed in Table B-6. The equation used to calculate o_z is of the form:

$$o_z = ax^b \qquad \text{Equation (B-6)}$$

where the downwind distance x is in kilometers and o_z is in meters in Equation (B-6) and the coefficients a and b are given in Table B-7.

Tables B-8 and B-9 show the equations used to determine o_y and o_z for Urban Mode 3. These expressions were determined by Briggs as reported by Gifford[25] and represent a best fit to urban vertical diffusion data reported by McElroy and Pooler.[26] The Briggs functions are assumed to be valid for downwind distances less than 100 m. However, concentrations at receptors less than 100 m from a source may be suspect.

B.3.3 Vertical Term

The Vertical Term includes the effects of source elevation, plume rise, limited mixing in the vertical, and the gravitational settling and dry deposition of particulates. The Vertical Term is defined as follows:

$$V_d = [\bar{b}H + (1 - \bar{b}) H_v] \exp [-0.5((H - H_v)/o_z)^2] +$$
$$\sum_{i=1}^{\infty} (B_1B_2 + B_3B_4) \qquad \text{Equation (B-7)}$$

where:

y = reflection coefficient
$\bar{b}$ = average value of exponent b for the interval between the source and the downwind distance x (see Tables B-6 through B-9)
$H = h + \Delta h$
h = stack height (m)
Δh = plume rise (m)
$H_v = V_{sn} x/u$
V_{sn} = settling velocity of particulates in the nth settling-velocity category (m/s)

TABLE B-6. PARAMETERS USED TO CALCULATE PASQUILL-GIFFORD σ_y[a]

Pasquill Stability Category	σ_y (meters) = 465.11628 (x) tan (TH) TH = 0.017453293 (c - d ln x)	
	c	d
A	24.1670	2.5334
B	18.3330	1.8096
C	12.5000	1.0857
D	8.3330	0.72382
E	6.2500	0.54287
F	4.1667	0.36191

[a] Where σ_y is in meters and x is in kilometers.

TABLE B-7. PARAMETERS USED TO CALCULATE PASQUILL-GIFFORD σ_z

Pasquill Stability Category	x (km)	σ_z (meters) = $a x^b$	
		a	b
A[a]	<.10	122.800	0.94470
	0.10 - 0.15	158.080	1.05420
	0.16 - 0.20	170.220	1.09320
	0.21 - 0.25	179.520	1.12620
	0.26 - 0.30	217.410	1.26440
	0.31 - 0.40	258.890	1.40940
	0.41 - 0.50	346.750	1.72830
	0.51 - 3.11	453.850	2.11660
	>3.11	[b]	[b]
B[a]	<.20	90.673	0.93198
	0.21 - 0.40	98.483	0.98332
	>0.40	109.300	1.09710
C[a]	All	61.141	0.91465
D	<.30	34.459	0.86974
	0.31 - 1.00	32.093	0.81066
	1.01 - 3.00	32.093	0.64403
	3.01 - 10.00	33.504	0.60486
	10.01 - 30.00	36.650	0.56589
	>30.00	44.053	0.51179
E	<.10	24.260	0.83660
	0.10 - 0.30	23.331	0.81956
	0.31 - 1.00	21.628	0.75660
	1.01 - 2.00	21.628	0.63077
	2.01 - 4.00	22.534	0.57154
	4.01 - 10.00	24.703	0.50527
	10.01 - 20.00	26.970	0.46713
	20.01 - 40.00	35.420	0.37615
	>40.00	47.618	0.29592
F	<.20	15.209	0.81558
	0.21 - 0.70	14.457	0.78407
	0.71 - 1.00	13.953	0.68465
	1.00 - 2.00	13.953	0.63227
	2.01 - 3.00	14.823	0.54503
	3.01 - 7.00	16.187	0.46490
	7.01 - 15.00	17.836	0.41507
	15.01 - 30.00	22.651	0.32681
	30.01 - 60.00	27.074	0.27436
	>60.00	34.219	0.21716

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[a] If the calculated value of σ_z exceeds 5000 m, σ_z is set to 5000 m.
[b] σ_z is equal to 5000 m.

TABLE B-8. BRIGGS FORMULAS USED TO CALCULATE McELROY-POOLER σ_y

Pasquill Stability Category	σ_y (meters)[a]
A	$0.32\ x\ (1.0 + 0.0004\ x)^{-1/2}$
B	$0.32\ x\ (1.0 + 0.0004\ x)^{-1/2}$
C	$0.22\ x\ (1.0 + 0.0004\ x)^{-1/2}$
D	$0.16\ x\ (1.0 + 0.0004\ x)^{-1/2}$
E	$0.11\ x\ (1.0 + 0.0004\ x)^{-1/2}$
F	$0.11\ x\ (1.0 + 0.0004\ x)^{-1/2}$

[a] Where x is in meters.

TABLE B-9. BRIGGS FORMULAS USED TO CALCULATE McELROY-POOLER σ_z

Pasquill Stability Category	σ_z (meters)[a]
A	$0.24\ x\ (1.0 + 0.001\ x)^{-1/2}$
B	$0.24\ x\ (1.0 + 0.001\ x)^{-1/2}$
C	$0.20\ x$
D	$0.14\ x\ (1.0 + 0.0003\ x)^{-1/2}$
E	$0.08\ x\ (1.0 + 0.0015\ x)^{-1/2}$
F	$0.08\ x\ (1.0 + 0.0015\ x)^{-1/2}$

[a] Where x is in meters.

u = wind speed (m/s)
x = downwind distance (m)
$B_1 = y^{1-1} [\bar{b}H_1 - (1-\bar{b}) H_v]$
$B_2 = \exp [-0.5(H_1 + H_v)/o_z)^2]$
$B_3 = y^1 [\bar{b} H_2 + (1-\bar{b}) H_v]$
$B_4 = \exp [-0.5 (H_2 - H_v)/o_z)^2]$
$H_1 = 2iH_m - H$
$H_2 = 2iH_m + H$
H_m = mixing height (m)

B.3.4 Reflection Coefficient

To calculate the Vertical Term, information about the particle size distribution and the density of the particulates emitted must be known. Total particulate emissions are subdivided into categories and the gravitational settling velocity is calculated for the mass-mean diameter of each category. The gravitational settling velocity for all sizes of particulates is calculated according to the following equation by McDonald:[27]

$$V_s = 2pgr^2/9u \qquad \text{Equation (B-8)}$$

where:

V_s = settling velocity ($\text{cm} \bullet \text{sec}^{-1}$)
p = particle density ($\text{gm} \bullet \text{cm}^{-3}$)
g = acceleration due to gravity ($980\ \text{cm} \bullet \text{sec}^{-2}$)
r = particle radius (cm)
u = absolute viscosity of air ($u - 1.83 \times 10^{-4}\ \text{gm} \bullet \text{cm}^{-1} \bullet \text{sec}^{-1}$)

It should be noted that the settling velocity calculated using Equation (B-8) must be converted by the user from centimeters per second to meters per second for use in the model calculations.

The reflection coefficient y_n can be estimated for each particle-size category using Figure B-1 and the settling velocity calculated for the mass-mean diameter. If it is desired to include the effects of

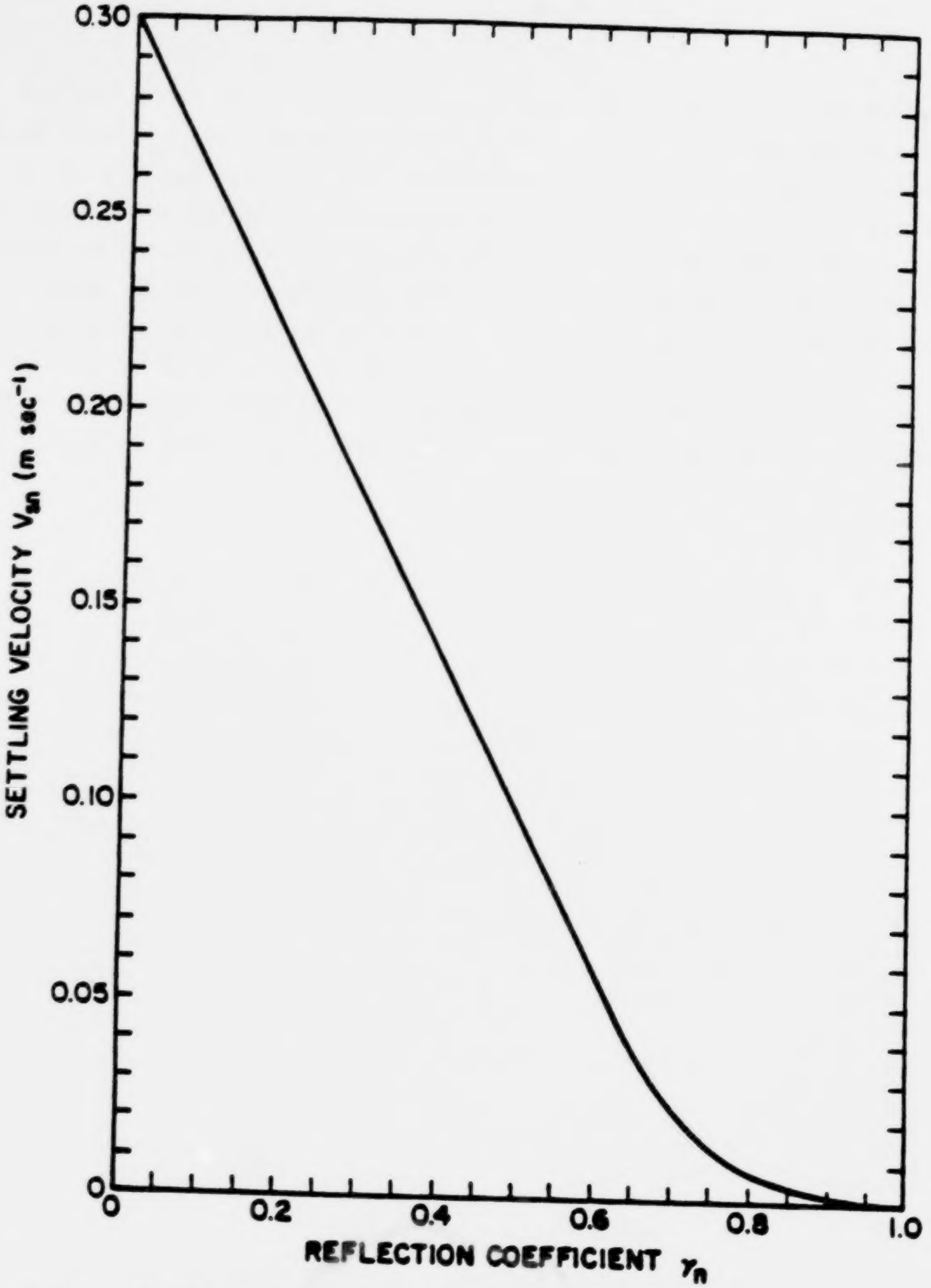

Figure B-1. Relationship between the Gravitational Settling Velocity V_{sn} and the Reflection Coefficient y_n suggested by Dumbauld, et al.[28]

gravitational settling in calculating ambient particulate concentrations while at the same time excluding the effects of deposition, y_n should be set equal to unity for all settling velocities. On the other hand, if it is desired to calculate maximum possible deposition, y_n should be set equal to zero for all settling velocities. The effects of dry deposition for gaseous pollutants may be estimated by setting the settling velocity V_{sn} equal to zero and the reflection coefficient y_n equal to the amount of material assumed to be reflected from the surface. For example, if 20 percent of a gaseous pollutant that reaches the surface is assumed to be retained at the surface by vegetation uptake or other mechanisms, y_n is equal to 0.8.

B.4 REFERENCES FOR APPENDIX B

1. U. S. Environmental Protection Agency. User's Manual for Single Source (CRSTER) Model. EPA-450/2-77-013. 1977.
2. Briggs, G. A. Plume Rise. USAEC Critical Review Series. NTIS Publication No. TID-25075. 1979.
3. Briggs, G. A. Some Recent Analyses of Plume Rise Observations. Proceedings of the Second International Clean Air Congress. 1971.
4. Briggs, G. A. Discussion on Chimney Plumes in Neutral and Stable Surroundings. Atmos. Environ. 6:507-510. 1972.
5. Briggs, G. A. Diffusion Estimates for Small Emissions. USAEC Report ATDL-106. 1974.
6. Briggs, G. A. Plume Rise Predictions. In: Lectures on Air Pollution and Environmental Impact Analyses. American Meteorology Society. 1975.
7. Huber, A. H., and W. H. Snyder. Building Wake Effects on Short Stack Effluents. Preprint Volume for the Third Symposium on Atmospheric Diffusion and Air Quality. American Meteorological Society. 1976.
8. Huber, A. H. Incorporating Building/Terrain Wake Effects on Stack Effluents. Preprint Volume for the Joint Conference on Applications of Air Pollution Meteorology. American Meteorology Society. 1976.
9. Pasquill, F. Atmospheric Dispersion Parameters in Gaussian Plume Modeling. Part II. Possible Requirements for Change in the Turner Workbook Values. EPA-600/4-76-030b. 1976.
10. Reference 2.
11. Reference 3.
12. Reference 4.
13. Reference 5.
14. Reference 6.
15. Reference 7.
16. Reference 8.
17. Turner, D. B. Workbook of Atmospheric Dispersion Estimates. PHS Publication No. 999-AP-26. U. S. Department of Health, Education and Welfare, National Air Pollution Control Administration. 1970.

18. Dumbauld, R. K., J. E. Rafferty, and H. E. Cramer. Dispersion - Deposition from Aerial Spray Releases. Preprint Volume for the Third Symposium on Atmospheric Diffusion and Air Quality. American Meteorological Society. 1976.

19. Cramer, H. E., et al. Development of Dosage Models and Concepts. Final Report. U. S. Army, Desert Center Report DTC-TR-609. 1972.

20. Reference 1.

21. Catalano, J. A. Single-Source (CRSTER) Model, Addendum to the User's Manual. U. S. Environmental Protection Agency. 1986.

22. Gifford, F. A. Turbulent Diffusion - Typing Schemes: A Review. Nucl. Saf. 17:68-86. 1976.

23. McElroy, J. L., and F. Pooler. The St. Louis Dispersion Study. U. S. Public Health Service - National Air Pollution Control Administration Report AP-53. 1968.

24. Reference 17.

25. Reference 22.

26. Reference 23.

27. McDonald, J. E. An Aid to Computation of Terminal Fall Velocities of Spheres. J. Met. 17:463. 1960.

28. Reference 18.

APPENDIX C
WET DEPOSITION MODEL

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WET DEPOSITION MODEL

This appendix presents the approach developed by EPA to model the effects of wet deposition on municipal waste combustor emissions.[1] The algorithm developed to predict wet deposition was combined with the existing ISCST dry deposition model to predict total surface deposition of municipal waste combustor emissions.

The following equation was used by EPA to model wet deposition:

$$wDep(n) = f\,\frac{\Lambda(n,j)\ \Phi n K Q_{\tau}}{\sqrt{2\pi}\ \sigma_y\ \bar{u}(h)}\exp\left[-\ \Lambda(n,j)\ \left(\frac{x}{\bar{u}(h)}\right)\right]\exp\left[-\ \frac{1}{2}\ \left(\frac{y}{\sigma_y}\right)^2\right] \quad \text{Equation (C-1)}$$

where:

f = fraction of time precipitation occurs (fraction of hour)
$\Lambda(n,j)$ = the fraction of material removed per unit time in the nth particle size category and jth precipitation intensity category
$\Phi(n)$ = fraction of total source material in the nth particle size category
K = units conversion factor
Q_{τ} = total source emission strength in grams
σ_y = standard deviation of the crosswind deposition distribution in meters
$\bar{u}(\underline{h})/x$ = the travel time of the plume in seconds
$u(h)$ = wind speed at stack height h in meters per second
x = alongwind distance from source to receptor in meters
y = crosswind distance from source to receptor in meters

The relationship between the scavenging coefficient and the particle size and precipitation intensity were derived from Radke et al.[2] Table C-1 summarizes the scavenging coefficients assigned to particle size categories and precipitation events used for computing estimates of wet deposition. The value, f, in Equation C-1 is applied to each hour of reported precipitation as determined from National Weather Service meteorological observations. A factor of 0.25 representing a 15-minute duration of precipitation was assigned to those hours influenced by thunderstorms or

TABLE C-1. SUMMARY OF SCAVENGING COEFFICIENTS EXPRESSED PER SECOND OF TIME, USED IN COMPUTING WET SURFACE DEPOSITION

Precipitation Intensity[a]	Particle Size Category (microns)		
	≤2	>2 and ≤10	>10
Heavy	1.46 E-3	4.64 E-3	9.69 E-3
Moderate	5.60 E-3	8.93 E-4	9.69 E-3
Light	2.20 E-4	1.80 E-4	9.69 E-3

[a] Light = less than 0.10 inches per hour.
Moderate = 0.11 to 0.30 inches per hour.
Heavy = equal to or greater than 0.31 inches per hour.

squalls; a factor of 0.5 was assigned to represent a 30-minute duration of precipitation for hours experiencing showers, and a value of 1.0 (60-minute duration) was assigned for hours with steady precipitation. In computing the dry deposition which occurs between the period of precipitation, a factor of (1-f) was used to determine the fraction of the material emitted during the hour that is subject to dry deposition. Thus, no dry deposition occurs during hours of steady precipitation.

Few published reports with data on the scavenging coefficients for snow could be found. Graedel and Franey have computed scavenging coefficients of 0.5 micron size aerosol particles in the range of 18 to 28 x 10^{-5} per second.[3] Wolf and Dana report an intensity dependent rate of 0.5 x 10^{-5} per second times the precipitation intensity for 0.5 micron particles, and have assumed this to define the lower bound coefficients for light and moderate snowfall.[4] However, it is not clear from the literature what the scavenging coefficients for snowfall would be for the particle size categories used in this study. Thus, in the wet deposition calculations for this analysis, EPA has used the same washout coefficients for both rainfall and snowfall events. Other frozen forms of precipitation, e.g., snow pellets, ice pellets, and hail, present a relatively smooth surface to the aerosol particles and are not likely to be effective scavengers of the pollutants. Therefore, periods and occurrences of these types of frozen precipitation are modeled in the dry deposition mode.

The principal assumptions made in computing wet deposition are: (1) the intensity of precipitation is constant over the entire path between the source and receptor; (2) the precipitation originates at a level above the top of the emission plume so that the hydrometers pass vertically through the entire plume, (3) the time duration of the precipitation over the entire path between the source and receptor point is such that exactly f, a faction between zero and one, of the hourly emission Q in Equation (C-1) is subject to a constant intensity for the entire travel time required to traverse the distance between the source and receptor, and the remaining fraction (1-f) is subject only to dry deposition processes.

C.1 REFERENCES FOR APPENDIX C

1. U. S. Environmental Protection Agency. Methodology for Assessment of Health Risks Associated with Multiple Pathway Exposure with Multiple Pathway Exposure to Municipal Waste Combustor Emissions. Submitted for Review to the Science Advisory Board. October 1986.

2. Radke, L. F., P. V. Hobbs, and M. W. Eltgroth. Scavenging of Aerosol Particles by Precipitation. Journal of Applied Meteorology 19:715-722. 1980.

3. Graedel, T. E., and J. P. Franey. Field Measurements of Sub-Micron Aerosol Washout by Rain. Proceedings of the Symposium on Precipitation Scavenging. ERDA Symposia Series 41:503-523. 1977.

4. Wolf, M. A., and M. T. Dana. Experimental Studies on Precipitation Scavenging. Battelle Northwest annual report. USAEC Report BNWL-1051 (Part 1), pp. 18-25. 1969.

APPENDIX D
TERRESTRIAL FOOD CHAIN MODEL

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D.1 INTRODUCTION

Contaminants associated with emissions from municipal waste combustors are subject to deposition on surfaces downwind from the combustor. The fallout may be deposited on soil and/or vegetation. The Terrestrial Food Chain Model (TFC) applies to both the human food chain and the ecological food chain. The pathways in the model assess exposure to humans, animals, soil biota and vegetation. Terrestrial trophic relationships are numerous and complex. Humans in the vicinity of municipal waste combustors have the potential to ingest contaminated soil directly or consume vegetation and animal tissues containing the contaminants. Herbivorous animals, plants, and soil organisms living where municipal waste combustor emissions are deposited may also be exposed. Humans or other organisms may be exposed to the soil-deposited contaminants of municipal waste combustor emissions by several pathways. For each exposure pathway, it is important to identify the most-exposed individual organism, or MEI. The MEI is a hypothetical individual or organism, but care should be taken that the definition is realistic. The definition of the MEI will vary with each pathway.

In the case of humans, occupational exposures are not considered. It is assumed that workers involved in the operation of municipal waste combustors can be required to use special measures or equipment to minimize their exposure to possibly hazardous materials. This methodology is geared toward protection of the general public and the environment. While many individuals of the general public may be exposed to a varying degree, the MEI is that individual who would be expected to experience the greatest risk and, therefore, require the greatest protection. For the pathways involving ecological effects, the MEI is a sensitive plant or animal rather than a human.

The remainder of this appendix describes the MEI assumed for each exposure pathway, the approach for estimating soil concentration and contaminant uptake in the organisms of interest, and the assumptions, calculation steps, and input requirements for estimating exposure for each pathway. Example calculations of exposure estimates are also given for each pathway.

D.2 MOST EXPOSED INDIVIDUAL (MEI) ASSUMED FOR EACH EXPOSURE PATHWAY

D.2.1 Crops for Human Consumption

This pathway (deposition-soil-plant-human toxicity) is important wherever crops for human consumption are grown in a vicinity where emissions from a municipal waste combustor may be deposited. Uptake of the deposited contaminants is assumed to occur through the plant roots. Direct adherence of deposited contaminants or soil to crop surfaces is not considered here. (Direct ingestion of deposited contaminants is discussed in Section D.2.2.) The MEI is defined as an individual residing in a region within 50 km of a municipal waste combustor in the area of maximal deposition of emissions. The individual who grows a large proportion of his or her own food would result in highest risk since much of the diet would be potentially affected.

D.2.2 Soil Ingestion by Children

Human adults may ingest some soil, but the amounts consumed by young (i.e., preschool, 1-6 years of age) children are much greater. This is especially true in children exhibiting the behavior known as "pica," the ingestion of nonfood items; these children constitute the MEI for this pathway (deposition-soil-human toxicity). Preschool children with pica for soil are assumed to be exposed in residential areas within 50 km of the municipal waste combustor in the areas of maximal deposition of emissions. The exposure is likely to occur in gardens, lawns, landscaped areas, parks and recreational areas.

D.2.3 Herbivorous Animals for Human Consumption

Two separate pathways are considered whereby animal products may become contaminated: 1) deposition-soil-plant-animal-human toxicity and 2) deposition-soil-animal (direct ingestion)-human toxicity. By the first pathway, row crops (i.e., grains) or other forage crops (i.e., grasses) are grown on soils contaminated by municipal waste combustor emissions and take up contaminants through the roots. The crops are then harvested for animal consumption. By the second pathway, the deposited contaminants adhere to crop surfaces or remain in the thatch layer on the soil surface. The crop is then harvested or grazed, resulting in ingestion of deposited particles or pollutants. In addition to domestic grazers, wild herbivores, such as deer, may forage grains or grasses within the range of emissions fallout and be taken by hunters. The MEI for this pathway is the human consumer of these animal products.

D.2.4 Toxicity to Herbivorous Animals

The pathways for exposure for herbivorous animals are 1) deposition-soil-plant-animal toxicity, and 2) deposition-soil-animal toxicity (direct ingestion). For these pathways, it does not matter whether the animals are subsequently consumed by humans. The end point of concern is toxicity to the animals themselves, which constitute the MEI. It is assumed that wildlife may forage on lawns, gardens, agricultural areas or forests within the region of maximal deposition of emissions of the MWC.

D.2.5 Phytotoxicity

This pathway is described as deposition-soil-plant toxicity. Toxic effects in plants are of concern since plants have an integral role in the terrestrial food chain. The MEI, or vegetation type to be protected, will ordinarily be the most sensitive plan species for which data are available.

D.2.6 Toxicity to Soil Biota or Their Predators

Two pathways are considered here: 1) deposition-soil-soil biota toxicity, and 2) deposition-soil-soil biota-predator toxicity. The term "soil biota" is intended to be interpreted broadly. The first pathway examines effects on a broad range of organisms including microorganisms, soil invertebrates such as earthworms, and various anthropods living in or near the soil, as long as potential effects in these organisms can be related to soil concentrations. The second pathway examines exposures to predators of these organisms, especially small mammals and birds. These predators could include insectivores as long as available data permit the contaminant concentration in the prey to be related to contaminant soil concentration.

D.3 ESTIMATION OF SOIL CONCENTRATION AND CONTAMINANT UPTAKE BY ORGANISMS

D.3.1 Soil Deposition Rate of Contaminants

The cumulative soil deposition rate of contaminants (in kg/ha) is determined from the total (dry plus wet) deposition rate of the pollutant $[g/m^2\ (year)]^{-1}$ over the total period of deposition from the municipal e combustor by the following equation:

$$CD = AD \times T \times 10 \qquad \text{Equation (D-1)}$$

where:

CD = cumulative soil deposition of pollutant (kg/ha)
AD = annual deposition rate of pollutant $[g/m^2\ (year^{-1})]$
T = total period of deposition (years)
10 = conversion factor $[m^2 \times kg/(ha \times g)]$

For this methodology, the lifetime of the municipal waste combustor (or total period of deposition) is considered to be $\geq$30 years; however, since the site is already dedicated for municipal waste combustion, it may be assumed that the combustor will be replaced. Therefore, the lifetime of the municipal waste combustor could be as great as 100 years.

D.3.2 Soil Incorporation of Deposited Contaminants

Following deposition, contaminants from emissions of municipal waste combustors may be incorporated into the upper layer of soil where crops or other vegetation are grown. If incorporation is accomplished by disking or plowing of the upper layer of soil, it is assumed that the deposited pollutants would be mixed into the soil to a depth of ~20 cm (8 inches).

If the contaminants persist indefinitely in the upper soil layer, as is the case for some inorganics, the following relationship exists between contaminant deposition rates and concentration increment in soil:

$$LC = (CD \times 10) \times (B \times D)^{-1} \qquad \text{Equation (D-2)}$$

where:

LC = maximal soil concentration increment of pollutant (ug/g DW)
CD = cumulative soil deposition of pollutant (kg/ha)
10 = conversion factor $[(m^2 \times ha \times ug)/(cm^2 \times m^2 \times kg)]$
D = depth of soil layer (cm)
B = bulk density of soil (g/cm^2)

From Equation (D-2), a soil concentration of 1 ug/g DW corresponds to a deposition rate of 2.7 kg/ha. LC represents the concentration increment, not the total concentration, because it does not take into account background concentrations of the contaminant that may already be present, whether natural or from other pollution sources.

Where soil incorporation does not occur, particulate is assumed to be retained in a shallower, uppermost soil layer. While the actual depth of this uppermost layer retaining the unincorporated contaminant is unknown, a value of 1 cm will be assumed.

D.3.3 Contaminant Loss from Soils

Contaminants may be lost from soils as a result of numerous processes, including leaching, volatilization and chemical and biological degradation. These processes may occur simultaneously or at different rates.

Organic contaminants and inorganic contaminants may be subject to some or all of these loss processes; thus, it may be extremely difficult to model overall rate of loss. A simple means to estimate loss is based on empirical data from soil systems where soil concentrations have been followed over time. These data may be used to estimate a first-order loss rate constant for the pollutant. The use of such a rate constant is recognized to be an over-simplification since the processes involved are complex and not necessarily or only first-order. Where no basis for an estimate is available, no loss should be assumed. The maximal soil concentration of chemicals subject to loss (for all $k > 0$) may be calculated as a function of the annual deposition rate constant as shown in the equation below:

$$LC_T = AD \times (1 - e^{-kT}) \times 10^2 \times (B \times D \times k)^{-1} \qquad \text{Equation (D-3)}$$

where:

LC_T = maximal soil concentration of pollutant after time, T (ug/g DW)
AD = annual deposition of contaminant [g/(m^2 x year)]
MS = 2.7 x 10^3 Mg/ha = assumed mass of soil in upper 20 cm
k = loss rate constant $(\text{years})^{-1}$
T = total period of deposition (years)
10^2 = conversion factor [(m^2 x mg)/(cm^2 x g)]
B = bulk of density of soil (g/cm^2)
D = depth of soil layer (cm)

This formula is derived from an environmental application of toxicokinetic principles.[1]

D.3.4 Contaminant Uptake Relationship in Plants

D.3.4.1 Plant Uptake of Inorganics. Uptake rates of inorganic chemicals, especially uptake of metals by plants, have been recently reviewed.[2,3,4] Ryan et al. used linear regression (of plant tissue Cd concentration against applied Cd) to derive uptake response slopes; i.e., the increase in tissue concentration for various crops.[5] [The term "uptake response" is used to denote an increase in tissue concentration in response to exposure to a chemical; i.e., the difference in pre- and post-exposure concentrations.] These authors stated that although the uptake slope could be altered by several variables, it remained essentially linear.

More recent work has shown that plant response to metals from sludge-amended soil is curvilinear, approaching a plateau concentration in tissue as sludge application rate increases;[6] however, metal-adsorptive materials present in the sludge matrix are thought to be responsible for this effect. Since no such uptake limiting effect has been demonstrated for deposited metals, uptake response slopes will be assumed to be linear for this methodology. The assumption that response slopes are linear means that dietary intake of a contaminant increases continually with contaminant application (or deposition) to soil (if some or all of the diet originates from these soils). Ryan et al. further assumed that response was related to cumulative Cd application. Using this approach, a limit can be derived for the cumulative application of a metal based on its dietary threshold level in humans.[7]

Linear response slopes can be calculated from any data set where tissue analyses and cumulative metal deposition or application rates have been recorded, assuming that the metal is not significantly lost over time. Plant tissue contaminant concentration (in ug/g DW) is regressed against cumulative contaminant application or deposition (in kg/ha) for the various treatment levels, including the control, to calculate the uptake response slope. If contaminant concentrations in soil, LC, were measured rather than deposition or application rates, the cumulative deposition, CD, may be calculated based on Equation (D-2). For inorganics that are lost over time, however, tissue concentration should be regressed against LC.

Wherever possible, uptake response data derived from areas of emission fallout will be used. For contaminants lacking such data, linear uptake slopes derived from other types of chemical application (such as sludge or pesticide additions) will be assumed to apply to deposited contaminants as well.

If the contaminant is phytotoxic, a maximum tissue concentration for a given crop will be determined based on available phytotoxicity data, and assumed as an upper limit to uptake. Phytotoxicity of metals may be altered by soil pH. Phytotoxicity data chosen should be appropriate for the soil pH of the fallout region of the municipal waste combustor if possible. Maximum concentrations are those associated with severe yield reduction (>75%) or death of the plant, which would preclude pollutant passage up the foodchain.

D.3.4.2 Plant Uptake of Organics. Linear uptake response is also assumed for organic chemicals and is calculated as described for inorganics, but with some important differences. Because organic compounds and some inorganics tend to degrade in soil, plant tissue concentration is usually expressed as a function of a measured soil concentration, rather than application or deposition rate, for chemicals subject to loss. Therefore, the soil concentration (in ug/g DW) is regressed against tissue concentration to determine the uptake slope.

In addition, because soil concentration rather than deposition rate is used, and because most of the compounds of concern are xenobiotics, tissue concentration can be assumed to be zero when soil concentration is zero. Therefore, the slope reduces to a bioconcentration factor that can be derived from a single data pair.

D.3.5 Contaminant Uptake by Animal Tissues

Linear response slopes are derived for uptake of inorganics or organics by animal tissues consumed by humans. Tissue concentration is regressed against concentration in feed. Tissue concentrations in the literature may be expressed in dry or wet weight, but dry weight is preferred. For uniformity in applying this methodology, all slopes should be derived based on dry weight (moisture free, but including fat) concentrations in tissue and feed. Conversion from wet to dry weight for various tissues should be made according to percent moisture values given in USDA (1975) or another authoritative source.

For lipophilic organics, tissue concentration is often expressed on a fat basis (ug/g fat). If so, the uptake slope should be expressed on a fat basis rather than converted to a dry weight basis. Also, the slope for organics may be the same as a bioconcentration factor derived for a single data point (i.e., animal tissue concentration and feed concentration), as described previously for plant uptake of organics (see Section D.3.4.2).

D.3.6 Human Diet

Humans may be exposed to crops or animal products that have taken up the pollutants through the soil or diet, respectively. In order to quantify potential dietary exposures, it is necessary to estimate the amounts of various types of foods in the human diet. The most up-to-date and detailed source of information regarding food consumption habits of the United States population is the FDA Revised Total Diet Study food list.[8] While this food list provides a very detailed picture of the United States diet, it cannot be used in its published form for risk assessments of the present type. Many of the food items listed are complex prepared foods (such as soup or pizza), rather than the raw commodities (such as crops or meats) for which contaminant uptake data are available. Therefore, the reanalysis of the Pennington diet used recently by EPA will also be used in this methodology.[9] Each item in the Pennington diet (including the infant/junior foods) was broken down into its component parts, based on information available from FDA and USDA.[10,11] The percentages of dry matter and fat for each component were also specified. These components were then aggregated into the specific commodity groups required for this methodology. A summary of consumption for each category by each age/sex group is presented in Table D-1. This analysis should be considered preliminary as it has not been reviewed by the FDA.

D.4 ASSUMPTIONS AND CALCULATION STEPS FOR ESTIMATING EXPOSURE

D.4.1 Deposition-Soil-Plant-Human Toxicity Exposure Pathway

D.4.1.1. Assumptions. In addition to the assumptions listed in Table D-2, assumptions on the percent of diet affected by deposition of contaminants from municipal waste combustor emissions are made for this pathway. These assumptions and their potential limitations are summarized in Table D-3.

TABLE D-1. AVERAGE DAILY DRY-WEIGHT CONSUMPTION OF FOOD GROUPS, BASED ON A REANALYSIS OF THE FDA REVISED TOTAL DIET FOOD LIST

Food Group subgroup	Consumption by Age-Sex Group (g dry weight/day)							
			14-16 Years		25-30 Years		60-65 years	
	6-11 Months	2 Years	Females	Males	Females	Males	Females	Males
Grains and cereals								
wheat	42.969	61.820	81.980	124.475	69.534	102.207	65.114	89.525
corn	6.239	16.938	20.148	26.358	14.591	22.439	15.502	20.347
rice	3.003	4.586	5.137	6.638	4.776	6.783	4.024	4.396
oats	6.362	3.758	1.779	4.788	1.337	2.044	2.044	2.293
other grain	0.006	0.015	0.018	4.197	9.007	77.912	2.419	25.644
Potatoes	8.391	13.701	21.505	31.854	17.481	28.289	15.914	23.837
Leafy vegetables	0.838	0.482	1.141	1.306	2.176	2.164	2.783	2.526
Legume vegetables	2.475	4.683	6.459	10.651	7.987	11.891	8.454	11.098
Root vegetables	0.893	0.736	1.422	2.278	1.491	2.126	1.529	1.866
Garden fruits	0.900	2.001	3.766	4.448	4.446	5.943	4.797	5.442
Peanuts	0.243	1.661	1.374	3.162	1.311	2.840	1.115	2.137
Mushrooms	0.000	0.001	0.002	0.002	0.003	0.004	0.002	0.002
Vegetable oil	30.780	27.119	41.938	63.747	44.530	75.746	37.029	55.621
Meats								
beef	3.006	8.475	15.322	24.640	15.819	25.050	12.995	20.558
beef fat	11.885	8.974	14.783	23.702	15.167	27.952	12.175	18.619
beef liver	0.077	0.109	0.127	0.200	0.565	0.423	0.443	0.662
beef liver fat	0.1012	0.146	0.168	0.265	0.748	0.5596	0.5854	0.8762
lamb	0.0570	0.0314	0.0253	0.0180	0.1358	0.1064	0.0864	0.0873
lamb fat	0.127	0.070	0.056	0.040	0.302	0.237	0.192	0.194
pork	1.412	4.994	7.892	10.942	7.771	13.846	8.036	12.566
pork fat	4.869	8.264	10.556	15.698	10.590	20.046	10.554	16.582
poultry	2.253	4.515	6.997	8.258	6.607	9.681	6.736	8.169
poultry fat	5.925	1.412	1.602	1.809	1.454	2.061	1.464	1.824
fish (including fat)	0.339	1.200	2.540	2.676	3.501	4.670	3.629	4.108
Dairy	41.021	31.854	32.088	49.858	21.788	31.772	19.108	25.501
dairy fat	39.196	16.252	18.354	29.677	14.545	21.991	12.145	16.226
Eggs	3.395	6.788	5.998	8.103	6.286	9.197	7.000	10.468
Other	130.992	253.974	407.624	527.718	379.042	482.937	198.307	237.888

Source: References 9 and 10.

TABLE D-2. ASSUMPTIONS FOR TERRESTRIAL FOOD CHAIN

Functional Area	Assumptions	Ramifications/Limitations
Soil incorporation of contaminants	If soil incorporation is assumed, incorporation depth is 20 cm, and the upper 20 cm soil layer has a dry mass of 2.7 x 10^3 Mg/ha.	By this assumption, as soil concentration of 1 μg/g corresponds to a pollutant application of 2.7 kg/ha. If actual depth (and mass) is less, impacts on soil biota in the incorporation layer could be underpredicted (and vice versa). Ramifications for effects on plants is less clear, since less of the root zone is contaminated as incorporation depth decreases.
	Soil background concentration is not considered.	Effects could be underpredicted if background concentration is not considered.
Contaminant loss from soils	Trace metal contaminants are assumed to be conserved indefinitely in the upper layer unless loss constants are available.	Although most heavy metals are tightly bound to soil, measurements tend to show that this assumption often overpredicts concentrations and therefore, probably overpredicts certain hazards.
	Degradation of organic contaminants is first-order.	Could over- or underpredict degradation rate, which is complex and not necessarily first-order.
Plant uptake of inorganics	Uptake response slope may vary with pH.	Uptake response slope determined for one region may over- or underestimate uptake response slope for a region with different soil pH.
	Plant uptake is treated as linear with application rate until highly toxic concentrations in tissue are reached.	May overpredict tissue response to contaminant application if relationship is truly curvilinear.
	Plant uptake is treated as linear with soil concentration until highly toxic concentrations in tissue are reached.	May overpredict tissue response to contaminant application if relationship is truly curvilinear.
Animal uptake of contaminants	Animal tissue concentration is treated as a linear function of feed concentration.	Could under- or overpredict tissue concentration outside the observed range of response.
	Pure chemicals added to diet can be used to determine uptake response slope.	Availability of chemicals in soil may differ; especially, may be lower, so that uptake is overpredicted.
Human diet	The FDA Revised Total Diet Study food list (Pennington, 1983) is representative of the United States diet. The age/sex group with the highest consumption of a given crop group (typically the 25- to 30-year-old male) is the MEI for that item.	While very complete and detailed, the Pennington diet provides no information on variability within age/sex groups.

TABLE D-3. ASSUMPTIONS FOR DEPOSITION-SOIL-PLANT-HUMAN TOXICITY EXPOSURE PATHWAY

Functional Area	Assumptions	Ramifications/Limitations
Fraction of diet affected by deposition of MWC emissions.	All of an individual's homegrown food could come from soil with deposited emissions.	May overpredict exposure.
	The percentage of homegrown food in the diet of the MEI can be estimated from USDA survey data on rural farm households, which constituted 6 percent of all households.	More recent information if available, might show significant changes in both demographics and gardening habits of these households.

Source: Reference 12.

This exposure pathway deals with crops for human consumption. It will be assumed that home gardeners produce and consume leafy, legume and root vegetables, potatoes and garden fruits but not grains and cereals, peanuts or mushrooms. The USDA survey of United States food consumption in 1965-1966 includes data on the percentages of foods consumed that were homegrown for urban, rural nonfarm and rural farm households.[12] The highest percentages of homegrown foods were for rural farm households, which constituted ~6 percent of all United States households. The rural farm dweller in the area of maximal deposition within 50 km of a municipal waste combustor will be taken as the MEI in this pathway. All of the homegrown food by the MEI could come from soil contaminated with deposited emissions from the municipal waste combustor. The average percent of annual consumption that is homegrown for various foods from rural farm households, is shown in Table D-4.

D.4.1.2 Calculation Method. Uptake response slopes, in units of ug/g DW $(kg/ha)^{-1}$ for most inorganics or ug/g DW $(ug/g)^{-1}$ for organics or chemicals subject to loss, are used to determine dietary response to the deposited contaminant. The main disadvantage of using slopes is that the slopes for each crop will likely originate from different experimental conditions. In order to examine total dietary response to a change in conditions (such as soil pH), all of the response slopes may need to be changed. Lack of adequate uptake data for a chemical would preclude this calculation.

The following steps are taken to calculate daily intakes of contaminants by this pathway.

Step A. Sort Available Uptake Response Data for All Food Crops

For chemicals showing increased uptake at lower pH (i.e., many metals), the available response data for the crop should be grouped according to whether soil pH was <6.0 or >6.0. Studies with pH >6.0 should be used only if natural soils in the vicinity of the municipal waste combustor have a neural or alkaline pH.

TABLE D-4. AVERAGE PERCENT OF ANNUAL CONSUMPTION THAT IS HOMEGROWN FOR VARIOUS FOODS, RURAL FARMS HOUSEHOLDS

Food Group	Percent Homegrown
Milk, cream, cheese	39.9
Fats, oil	15.2
Flour, cereal	1.6
Meat	44.2
Poultry, fish	34.3
Eggs	47.9
Sugar, sweets	9.0
Potatoes, sweet potatoes	44.8
Vegetables (fresh, canned, frozen)	59.6
Fruit (fresh, canned, frozen)	28.6
Juice (vegetable, fruit)	11.0
Dried vegetables, fruits	16.7

Source: Reference 12.

Step B. Determine Uptake Response Slopes for Each Food Group

Response slopes with units of ug/g DW $(kg/ha)^{-1}$ or ug/g DW $(ug/g)^{-1}$, as appropriate, are determined for each crop food group shown in Table D-1. This slope may be determined as a weighted mean of all the available response slopes where weighting is according to the dry weight consumption of each crop. If the data do not permit determination of a weighted mean, an unweighted mean may be taken or, to be conservative, the highest single value may be chosen to represent that food group.

If the data indicate that a food group consists of some crops that are relatively high accumulators and some that are relatively low, it may be worthwhile to subdivide the food group on this basis. If no response slope can be determined for a particular crop food group, the highest value for any of the other groups should be assigned to that group.

Step C. Determine Human Daily Intake

The increase in DI (in ug/day) of an inorganic contaminant due to deposition of municipal waste combustor emissions by this pathway is calculated from the response data and deposition rate (see Section D.3.1) by the following equation:

$$DI = CD \times \sum_{i=1}^{n} (UC_i \times FC_i \times DC_i) \qquad \text{Equation (D-4)}$$

where:

DI = increment (above background) of daily intake of pollutants (ug/day)

CD = cumulative soil deposition of pollutant (kg/ha)

UC_i = uptake response slope for i^{th} food group [ug/g DW $(kg/ha)^{-1}$]

FC_i = fraction of i^{th} food group (crop) assumed to originate from contaminated soil (unitless)

DC_i = daily dietary consumption of i^{th} food group (g DW/day)

The derivation of DI by the deposition-soil-plant-human toxicity pathway for organic contaminants or chemicals subject to loss is largely the same as the procedure for inorganics based on the linear response model. One difference is that uptake data are not segregated on the basis of soil pH since no reason for doing this has been demonstrated. In addition, the procedure for calculating response slopes for organics differs somewhat from that for inorganics, in that tissue concentration is treated as a linear function of soil concentration rather than application or deposition rate, as explained in Section D.3.4.2. Thus, the maximal soil concentration, LC [from Equation (D-3)] of the contaminant is used in the above equation rather than CD for the determination of the DI for organics or chemicals subject to loss. The LC should be compared with the phytotoxicity threshold as described in Section D.3.4.1.

D.4.1.3 Input Parameter Requirements. The following individual parameters are required to calculate the human daily intake for the deposition-soil-plant-human toxicity pathway.

D.4.1.3.1 Fraction of food group assumed to originate from soil contaminated by municipal waste combustor emissions. All of the homegrown food is assumed to originate from contaminated soil, but homegrown food comprises ≤60 percent of the diet of the rural farm dweller (see Table D-4). Therefore, values of FC (after rounding) are 0.60 for all vegetables (except dried legumes), 0.45 for potatoes and 0.17 for dried legumes, based on the average percent of annual consumption of homegrown foods. Some food groups (such as mushrooms, peanuts and grains and cereals) are assumed to be unaffected and the FC is set at zero.

D.4.1.3.2 Uptake response slope (UC). Uptake response data are required for as many crops as possible in the food groups for which FC $\neq$ 0. If UC for the contaminants varies with soil pH, slopes appropriate to local soil pH should be chosen as discussed in Step A.

D.4.1.3.3 Daily dietary consumption of food group (DC). Values for DC (in g DW/day) are needed for each food group for which FC $\neq 0$. The values chosen should be appropriate for the MEI. Consumption data presented in Pennington and reanalyzed in Table D-1 are mean values for each of eight age/sex groups.[13] One could define the MEI in terms of the age/sex group having the highest consumption for all of the food groups combined or for each individual food group. Alternatively, one could estimate a 95th-percentile consumption level, based on variability of 1-day consumption; however, this procedure risks overestimation of long-term consumption.

D.4.1.4 Example Calculations. In this section, examples will be provided for this pathway using the metal cadmium, and the organic compound benzo(a)pyrene [B(a)P]. Examples will be calculated for the model municipal waste combustion facility assumed to be located in western Florida.

One of the most important steps in these calculations is the selection of values for each of the parameters involved. Value selection must be based on careful literature searches and evaluations of available data. For many parts of this methodology, value selection is a very data-intensive exercise.

Since the calculations presented in this section are only for the purpose of examples, the values employed will not be based on literature searches, but will rely on a few sources of readily available information. Therefore, the results do not constitute actual recommendations for risk assessment.

D.4.1.4.1 Cadmium.

Steps A and B. Sort Available Uptake Response Data for All Food Crops and Determine Uptake Response Slopes for Each Food Group

Studies are sorted according to soil pH. Natural soils in the vicinity of the southeastern United States would tend to have pH >6.0. The data of Dowdy and Larson (pH ≥ 6.0) will be used for this example in lieu of all of the available studies.[14] Response data on carrot and radish are used to derive a weighted mean response slope for the "root vegetable" food group. Consumption data are those for the 25- to 30-year-old male.[15]

Crop type	carrot	radish
Linear UC [ug/g $(kg/ha)^{-1}$)]	0.20	0.056
Dry weight consumption (g DW/day)	0.55	0.021

Example Calculation of Weighted Mean

$$UC_{\text{root veg.}} = \frac{(0.20 \times 0.55) + (0.056 \times 0.021)}{0.55 + 0.021}$$

$$= 0.19 \text{ ug Cd/g } [\text{kg Cd/ha}]^{-1}$$

Step C. Determine Human Daily Intake

CD for cadmium for 30 and 100 years of total deposition is calculated from Equation (D-1) using an AD = 1.088 x 10^{-2} g/m^2-year. The DI is determined using Equation (D-4). UC values for each food group have been derived from the data of Dowdy and Larson as illustrated above for root vegetables.[16] Values of DC for vegetables are obtained from Table D-1. DC values chosen are those for the age/sex group with highest consumption of that food group. Values of FC are those for the home garden scenario from Table D-4.

Food Group	UC	DC	FC	UC x DC x FC
Potatoes	0.038	31.85	0.45	0.545
Leafy vegetables	0.605	2.78	0.60	1.009
Legume vegetables, nondried	0.0053	3.38	0.60	0.011
Legume vegetables, dried	0.0053	8.51	0.17	0.008
Root vegetables	0.19	2.28	0.60	0.260
Garden fruits	0.073	5.94	0.60	0.260
Total				2.093

For T = 30 years, DI = CD (kg/ha) x 2.083 [ug x ha $(\text{kg x day})^{-1}$]
= 3.26 (kg/ha) x 2.093 [ug x ha $(\text{kg x day})^{-1}$]
= 6.83 ug/day

For T = 100 years, Di = 10.88 kg/ha x 2.093 [ug x ha $(\text{kg x day})^{-1}$]
= 22.77 ug/day

This represents the increase in daily human intake due to cadmium from municipal waste combustor emissions with total deposition of 30 or 100 years.

D.4.1.4.2 Benzo(a)pvrene

Steps A and B. Sort Available Uptake Response Data for All Food Crops and Determine Uptake Response Slopes for Each Food Group

Studies for organic compounds are not sorted according to soil pH. Using the data in Connor and consumption data for the 25- to 30-year-old male, the weighted mean response slope for the root vegetables are calculated previously, in the example for cadmium.[17,18] Leafy vegetable UC were taken from Connor.[19] Data for other food groups were not available and were assigned the highest available (i.e., root vegetables UC). Calculated values are shown in Step C below.

Step C. Determine Human Daily Intake

The DI is determined using Equation (D-4) with LC in place of CD. The DC values are obtained from Table D-1. DC values chosen are those for the age/sex group with highest consumption of that food group. Values of FC are those for the home garden scenario from Table D-4.

Food Group	UC	DC	FC	UC x DC x FC
Potatoes	1.74	31.85	0.45	24.939
Leafy vegetables	0.42	2.78	0.60	0.701
Legume vegetables, nondried	1.74	3.38	0.60	3.529
Legume vegetables, dried	1.74	8.51	0.17	2.517
Root vegetables	1.74	2.28	0.60	2.380
Garden fruits	1.74	5.94	0.60	6.201
Total				40.267

The maximum soil concentration of B(a)P is determined from the annual deposition using Equation (D-3). For B(a)P, the loss rate constant, k, estimated from data for biotic loss only is 0.16 $(\text{year})^{-1}$, and may therefore underestimate total soil loss.[20] The AD is 5.66×10^{-4} g/(m^2 x year). The bulk density, B, of 1.5 g/cm^3 for sandy clay loam is used.[21] For 30 years,

$$LC = 5.66 \times 10^{-4} \times [1-e^{-(0.16)(30)}] \times 10^{2} \times (1.50 \times 20 \times 0.16)^{-1}$$

$$= 1.18 \times 10^{-2}\ \text{ug/g}$$

For 100 years,

$$LC = 1.18 \times 10^{-2}\ \text{ug/g}$$

The LC is compared with the phytotoxicity threshold, however, but data are not available for benzo(a)pyrene. Therefore, for Equation (D-4) (as modified for organics or chemicals subject to loss):

For 30 and 100 years,

$$DI = 1.18 \times 10^{-2}\ \text{ug/g} \times 40.267\ \text{(g/day)} = 4.75 \times 10^{-1}\ \text{ug/day}$$

This represents the increase in daily human intake due to B(a)P from municipal waste combustor emissions.

D.4.2 Deposition-Human Toxicity ("Pica") Exposure Pathway

D.4.2.1 Assumptions. In addition to many of the assumptions listed in Table D-2, some additional assumptions are made for this pathway, relating to the degree of contaminated soil ingestion that could occur and the method of assessing potential effects. These assumptions and their potential ramifications are summarized in Table D-5 and are further discussed in the following sections.

D.4.2.2 Calculation Method. A human daily intake, DI (in ug/day DW), is determined as follows:

$$DI = LC_T \times I_s \qquad \text{Equation (D-5)}$$

TABLE D-5. ASSUMPTIONS FOR DEPOSITION-HUMAN TOXICITY ("PICA") EXPOSURE PATHWAYS

Functional Area	Assumptions	Ramifications/Limitations
Exposure assessment	Deposited emissions are not necessarily soil incorporated and may be concentrated in the uppermost soil layer. Deposited contaminant is assumed to be distributed within the uppermost 1 cm of soil, and ingested soil is assumed to originate from the same 1 cm layer.	Overestimates exposure in situations where soil incorporation occurs to any depth >1 cm. If incorporation is to a lesser depth, exposure is underestimated. For example, if exposure is to fallout dust directly, ingested soil could approach 100% deposited particulate.
	Deposition-contaminated soil may be ingested by children at the rates observed in studies of pica for soil.	Overestimates exposure to the extent the pica child frequents some areas where deposition has not occurred.
Effects assessment	It is assumed that pica may occur from 1-6 years of age. Cancer potency is adjusted to reflect a 5-year rather than 70-year exposure.	If the child is more susceptible to chemical carcinogenesis than the adult, a 5/70 adjustment could result in underestimation of hazard.

where:

I_s = soil ingestion rate (g DW/day)

LC_T = maximal soil concentration increment of pollutant after time, T (ug/g)

Deposited contaminant is assumed to be within the uppermost 1 cm of soil and ingested soil is assumed to originate from the same 1 cm layer. Soil concentration, LC, is therefore calculated according to Equation (D-2) for most inorganic chemicals or Equation (D-3) for chemicals that are subject to loss processes using 1.50 g/cm^3 as the bulk density and a depth, D, of 1 cm.

D.4.2.3 Input Parameter Requirements.

D.4.2.3.1 Soil ingestion rate (I_s). Soil ingestion has been recognized as an important source of exposure to several pollutants. For adults, a value of 0.02 g/day has been used to estimate ingestion.[22] Children may ingest soil by either inadvertent hand-to-mouth transfer or by intentional direct eating. Pica is the term for frequent, intentional eating of non-food objects. Lepow et al. estimated that children frequently mouthing their hands may inadvertently ingest $\geq$100 mg of soil/day.[23] Children who eat soil directly may ingest as much as 5 g/day,[24] thus establishing a plausible typical-to-worse range of 0.1 to 5 g/day.

Studies aimed at more accurately determining the range of ingestion rates have yielded some data, but are as yet inconclusive. Binder et al. conducted a pilot study to establish methods for determining soil ingestion rates in children living near a lead smelter.[25] Based on these preliminary data and the value used previously by EPA, a value of 0.5 g/day is suggested as a reasonably protective value for I_s. This value represents an estimate of the 95th percentile of soil ingestion in this study population.

D.4.2.4 Example Calculations.

D.4.2.4.1 Cadmium. The DI for cadmium is determined using Equation (D-5), with I_s = 0.5 g/day. The LC for cadmium calculated according to Equation (D-2) using a soil depth of 1 cm is 21.76 and 72.53 ug/g for 30 and 100 years, respectively. For 30 years,

$$DI = 21.76 \text{ ug/g} \times 0.5 \text{ g/day}$$
$$= 10.88 \text{ ug/day}$$

For 100 years,

$$DI = 72.53 \text{ ug/g} \times 0.5 \text{ g/day} \times 1$$
$$= 36.27 \text{ ug/day}$$

D.4.2.4.2 Benzo(a)pyrene. The organic compound B(a)P is a carcinogen. For this example, I_s is 0.5 g/day and LC is 2.36×10^{-1} ug/g for both 30 and 100 years of total deposition. Therefore, the DI is:

$$DI = 2.36 \times 10^{-1} \text{ ug/g} \times 0.5 \text{ g/day}$$
$$= 1.18 \times 10^{-1}$$

D.4.3 Exposure Pathways for Herbivorous Animals for Human Consumption

D.4.3.1 Assumptions. Animal forage may be contaminated by uptake through the plant roots of deposited pollutants (deposition-soil-plant-animal-human toxicity) or by adherence to plant surfaces or roots [deposition-soil-animal (direct ingestion)-human toxicity] of deposited particulate or contaminated soil. In both pathways, humans are exposed by consuming animal tissue, which has taken up the contaminants. In the second pathway, soil incorporation is not assumed and direct ingestion of the contaminant by farm animals may occur. Direct ingestion is also possible by animals such as deer that will be taken by hunters. The amount of game consumed by hunters, however, is assumed not to exceed the consumption of home-produced meat by farm dwellers. Therefore, the farm dweller is taken as the MEI for both pathways. Protection of the MEI is assumed to be protective of hunters as well. Additional assumptions are listed in Table D-6.

TABLE D-6. ASSUMPTIONS FOR PATHWAYS DEALING WITH HERBIVOROUS ANIMALS

Functional Area	Assumptions	Ramifications/Limitations
Fraction of human diet affected by MWC emissions	The MEI is assumed to be an individual raising much of his own meats, poultry, eggs and dairy products. The percentage of home-produced foods in the diet of the MEI is estimated from a USDA (1966) survey of rural farm households, which constituted 6% of all households.	More recent information, if available, might show significant changes in both demographics and food production habits of these households.
	Individuals consuming wild game that forage in emissions-contaminated areas are assumed to have no greater exposure than the MEI identified above.	Information to substantiate this assumption is not immediately available.
Ingestion by animals of particulate-contaminated soil or forage crops	Contaminant uptake by crops may affect all animal feeds, but only grazing animals are affected by adherence. Adulteration by soil of harvested crops such as grains fed to non-grazing animals is assumed to be minimal.	
	Deposited contaminant is assumed to be distributed within the upper most 1 cm of soil, and ingested soil is assumed to originate from the same 1 cm layer. Direct ingestion of soil may occur, and only animals consuming pasture crops are affected.	
	The linear response slope of the most responsive forage crop is used to represent all forage crops in the animal diet.	Will tend to overpredict average crop response.

D.4.3.2 Calculation Method.

D.4.2.2.1 Uptake pathway: deposition-soil-plant-animal-human toxicity. To determine the human daily intake of a contaminant by this pathway, an animal feed concentration increment of the contaminant (AFC, in ug/g DW) first must be determined. The equations differ for organics and inorganics since the units of crop uptake differ. The AFC is calculated from the deposition rate [for inorganics, Equation (D-1) or the soil concentration [for organics or chemicals that are subject to loss, Equation (D-2)] and the crop uptake slope as follows:

(for most organics) $AFC = CD \times UC$ Equation (D-6)

(for chemicals subject to loss) $AFC = LC_T \times UC$ Equation (D-7)

where:

CD = cumulative deposition of pollutant (kg/ha)
UC = linear response slope of forage crop [ug/g crop DW $(kg/ha)^{-1}$]
or
[ug/g DW $(ug/g)^{-1}$]
LC_T = maximal soil concentration of pollutant, after time, T (ug DW)

Following the calculation of the AFC, the human daily intake (DI ug/day) is calculated as follows;

$$DI = AFC \times \sum_{i=1}^{n} (UA_i \times FA_i \times DA_i) \qquad \text{Equation (D-8)}$$

where:

AFC = animal feed concentration of pollutant (ug/g)
UA_i = uptake response slope of pollutant in the i^{th} animal tissue [ug/g tissue DW (ug/g feed DW)$^{-1}$]
DA_i = daily dietary consumption of the i^{th} animal tissue food group (g DW/day)

FA^i = fraction of i^{th} food group assumed to be derived from animals feeding on contaminated soil or feedstuffs

D.4.3.2.2 Adherence pathway: deposition-soil-animal (direct ingestion)-human toxicity. The animal feed concentration (AFC, in ug/g DW) is calculated as for the uptake pathway [see Equation (D-6)], but the values will differ because fewer human food categories will be affected. Soil incorporation is not assumed; therefore, the upper parts of pasture crops may be contaminated and may be harvested as feed for cattle or sheep or grazed directly by these animals. The deposited contaminant or particulate can also accumulate in the thatch layer of pasture, and be directly consumed by grazing animals. Domestic animals (such as pigs, poultry) that consume grains or other non-pasture crops are assumed not to be affected. When the deposited contaminant is consumed from plant or soil surfaces, contaminant intake by the grazing animal is related to the fraction of the animal diet which that soil comprises. Deposited contaminant is assumed to be within the uppermost 1 cm soil layer. Soil concentration (LC) is calculated according to Equation (D-2) for most inorganic chemicals or Equation (D-3) for chemicals which are subject to loss processes, using 1.50 g/cm^3 bulk density and a depth, D, of 1 cm. The animal feed concentration is derived in terms of the soil concentration and fraction of the animal diet that is adhering soil. AFC is derived as follows:

$$AFC = LC_T \times FS \qquad \text{Equation (D-9)}$$

where:

FS = fraction of animal diet that is adhering soil (unitless)
LC_T = maximal soil concentration increment of pollutant after time, T (ug/g)

D.4.3.3 Input Parameter Requirements.

D.4.3.3.1 Fraction of food group assumed to be derived from animals feeding on soil or feedstuffs contaminated by municipal waste combustor emissions (FA). As was the case with FC in the crops-for-human-consumption pathway (see Section D.4.1.3.1), this parameter determines which food groups are included in the analysis.

For the first of these two pathways (deposition-soil-plant-animal-human toxicity), all meat groups except fish will be assumed to be affected, including beef, lamb, pork, poultry, dairy products and eggs (see Table D-4). The second pathway [deposition-soil-animal (direct ingestion)-human toxicity] is assumed to affect only grazing animals (beef, lamb and dairy food groups are included).

As for the pathways dealing with crops for human consumption, the MEI for these two pathways is chosen as a farm family residing within 50 km of a municipal waste combustor (in the area of maximal contaminant deposition) raising a substantial percentage of their own meat or other animal products. The choice of FA values is based on the percentage of homegrown foods consumed by rural farm households (see Table D-4). As stated previously, it is assumed that these FA values are sufficiently high that an individual consuming wild game from contaminated areas will also be protected.

D.4.3.3.2 Daily dietary consumption of food group (DA). Values for daily dietary consumption (DA, in g DW/day) are needed for each food group for which FA $\neq$ 0. As was described for DC (see Section D.4.1.3.3), consumption data are taken from Table D-1. The food item "beef liver" includes various other organ meats consumed by humans in smaller amounts, such as kidney, hearts, etc. Individuals with a preference for those organs are expected to consume them at the rates given for beef liver. It is also assumed that consumption of wild game does not exceed the values of DA for other meats (beef, lamb) and will also protect hunters.

D.4.3.3.3 Fraction of animal diet that is adhering soil (FS). Studies of grazing animals indicate that soil ingestion ordinarily ranges from 1 to 10 percent of dry weight of diet (but may range as high as 20%) for cattle and may be $\geq$30 percent for sheep during winter months when forage is reduced.[27] It will be assumed that 100 percent of deposited contaminant is retained in the uppermost soil layer (~1 cm depth) that animals may ingest. Since lamb contributes relatively little to the United States diet, a value of FL = 10 percent or 0.10, based largely on cattle, will be used to represent a reasonably high exposure situation.

D.4.3.3.4 Uptake slope of contaminant in animal tissue (UA). Uptake slopes for affected tissues consumed by humans should be calculated as described in Section D.3.5. The form and availability of uptake response data will affect the types of food groups included in the calculation of AFC [see Equations (D-6) and (D-7)] and, therefore, the choice of values for FA and DA as well. If values of UA can be calculated for several different types of meats, then each of these meats can be separately included in Equation (D-8). If data for only a single type of meat are available, it may be necessary to assume that similar types of meat from different species have similar UA values. These tissues would then be grouped in Equation (D-8).

This analysis also assumes that UA values for wild game species are no greater than those for the domestic animals to which most available data apply. If data contradicting this assumption are available, these values may be used in Equation (D-8) to calculate DI.

D.4.3.3.5 Linear uptake response of forage crop (UC). The crop chosen for UC should be the one showing the highest response slope appropriate for the exposure scenario involved. Grains, legumes (such as soybeans), silage or grasses could be affected in agricultural use.

D.4.3.4 Example Calculations.

D.4.3.4.1 DI for uptake pathway.

Cadmium--The animal feed concentration of cadmium is first determined using Equation (D-6). The UC value of 0.14 ug CD/g (kg Cd/ha)$^{-1}$ based on corn silage will be used.[28] The CD for cadmium is 3.26 and 10.88 kg/ha for 30 and 100 years, respectively. Therefore, for 30 years,

$$\begin{aligned}\text{AFC} &= 3.26\ (\text{kg/ha}) \times 0.14\ \text{ug CD/g}\ (\text{kg Cd/ha})^{-1}\\ &= 0.46\ \text{ug/g}\end{aligned}$$

For 100 years,

$$\begin{aligned}\text{AFC} &= 10.88\ (\text{kg/ha}) \times 0.14\ \text{ug/g}\ (\text{kg Cd/ha})^{-1}\\ &= 1.52\ \text{ug/g}\end{aligned}$$

The DI for cadmium is calculated using Equation (D-8). Values of UA for various animal tissues were converted from wet to dry weight basis.[29] The value listed for "beef liver" is actually from sheep kidney; it is the highest UA value for an organ meat, since the beef liver consumption data are assumed here to represent any organ meat. No data were available for pork, so an average of the values for beef and lamb was used. Data were also unavailable for eggs and dairy products. In this example they are assumed to be similar to poultry and beef muscle, respectively. The DA values are derived from Table D-1. They include fat since the UA values are on a dry weight basis (including fat). The FA values are from Table D-4; they differ for the uptake and adherence pathways.

Animal Tissue Group	UA	DA	FA	UA x DA x FA (when FA ≠ 0)
Beef	0.003	53.0	0.44	0.070
Beef liver	9.9	1.54	0.44	6.71
Lamb	0.005	0.44	0.44	0.001
Pork	0.004	33.9	0.44	0.06
Poultry	0.08	11.7	0.34	0.32
Dairy	0.003	79.5	0.40	0.095
Eggs	0.08	8.1	0.48	0.31
Total				7.566

Therefore, the DI for cadmium for 30 years is,

$$DI = 0.45\ ug/g \times 7.566\ g/day = 3.48\ ug/day$$

For 100 years,

$$Di = 1.52\ ug/g \times 7.566\ g/day = 11.50\ ug/day$$

Benzo(a)pyrene--The animal feed concentration of B(a)P is determined using Equation (D-7). The UC value of 0.42 ug/g $(ug/g)^{-1}$ for spinach leaves was chosen since it had the greatest UC value of available foliage data.[30] For both 30 and 100 years,

$$\mathrm{AFC} = \mathrm{LC_T}\ (\mathrm{ug/g}) \times 0.42\ \mathrm{ug/g}\ (\mathrm{ug/g})^{-1}$$
$$= 4.96 \times 10^{-3}\ \mathrm{ug/g}$$

where LC_T (1.18×10^{-2} ug/g) is calculated based on a soil depth of 20 cm.

B(a)P is not extensively bioaccumulated in animals, making UA approximately zero.[31] Therefore, DI of B(a)P by the uptake pathway from consumption of animal tissue is not increased due to municipal waste combustor emissions.

D.4.3.4.2 Di for adherence pathway.

Cadmium--The AFC for this pathway is derived using Equation (D-9):

$$\mathrm{AFC} = \mathrm{LC_T} \times \mathrm{FS}$$

where:

LC_T = maximum soil concentration increment of pollutant after time, T (ug/g)

FS = fraction of animal diet adhering soil (unitless)

The value of FS will be 0.10 and the soil concentration of Cd (in 1 cm of uppermost soil layer) is 21.76 and 72.53 ug/g for 30 and 100 years, respectively.

For 30 years,

$$\mathrm{AFC} = 21.76\ \mathrm{ug/g} \times 0.10 = 2.18\ \mathrm{ug/g}$$

For 100 years,

$$\mathrm{AFC} = 72.53\ \mathrm{ug/g} \times 0.10 = 7.25\ \mathrm{ug/g}$$

Equation (D-8) and the UA and DA values from the uptake pathway are also used. The FA values from Table D-4 for meat and dairy (grazing animals only) are used.

Animal Tissue Group	UA	DA	FA	UA x DA x FA
Beef	0.003	53.0	0.44	0.070
Beef liver	9.9	1.54	0.44	6.71
Lamb	0.005	0.44	0.44	0.001
Pork	0.004	33.9	0	0
Poultry	0.08	11.7	0	0
Dairy	0.003	79.5	0.40	0.095
Eggs	0.08	8.1	0	0
Total				6.876

The DI for cadmium is 14.99 and 49.85 ug/day, for 30 and 100 years, respectively.

Benzo(a)pyrene--The AFC for this pathway is also calculated using Equation (D-9). The value of FS is 0.10 and the soil concentration of B(a)P in the uppermost 1 cm soil layer is 2.36×10^{-1} ug/g. For both 30 and 100 years,

$$AFC = 2.36 \times 10^{-1} \text{ ug/g} \times 0.10 = 2.36 \times 10^{-2} \text{ ug/g}$$

Since UA for B(a)P is approximately zero, DI calculated according to Equation (D-8) is zero, showing municipal waste combustor emissions do not increase DI of B(a)P by this pathway.

D.4.4 Exposure Pathways for Toxicity to Herbivorous Animals

These two pathways, deposition-soil-plant-animal toxicity and deposition-soil-animal toxicity (direct ingestion), are similar to the pathways for contaminant uptake by animal tissues consumed by humans, as discussed in Section D.4.3; however, since toxicity to the animal itself is now the endpoint of concern, the list of animals to be considered is broadened to include all herbivores. Herbivorous rodents and birds should be considered, as well as large herbivores and other domestic animals. The pathway for adherence of soils to plant roots is limited to grazing animals that ingest significant amounts of soil (cattle, sheep).

D.4.4.1 Assumptions. The assumptions pertaining to these pathways are stated in Tables D-1 through D-6.

D.4.4.2 Calculation Method. The first set of calculations for the deposition-soil-plant-animal and deposition-soil-animal pathways are the same as given in Section D.4.3.2 for the determination of an animal feed concentration (AFC, in ug/g DW) from the cumulative deposition (CD, for inorganics) or soil concentration (LC, for organics and/or chemicals subject to loss) and linear uptake response slope for crops (UC) [Equations (D-6), (D-7)]. The AFC is calculated for the deposition-soil-animal pathway from the soil concentration (LC) and fraction of soil constituting the animal diet (FS) as in Section D.4.3.2.2 [Equation (D-9)].

D.4.4.3 Input Parameter Requirements. The input parameters, LC and CD, were defined in Section D.3.2, UC was defined in Section D.4.1.3 and FS was defined in Section D.4.3.3.

D.4.4.4 Example Calculations.

D.4.4.4.1 Cadmium. The calculated AFC for cadmium for the uptake pathway, as described in Section D.4.3.4.1 (Cadmium) is 0.46 and 1.52 ug/g for 30 and 100 years, respectively. The AFC for the adherence pathway for cadmium is 2.18 and 7.25 ug/g for 30 and 100 years, respectively, calculated as described in Section D.4.3.4.2 (Cadmium).

For the uptake pathway (deposition-soil-plant-animal toxicity), a wider variety of herbivores may be affected than for the adherence pathway (deposition-soil-animal), which affects only grazing animals.

D.4.4.4.2 Benzo(a)pyrene. The AFC for B(a)P for the uptake pathway is calculated as 4.96×10^{-3} ug/g for both 30 and 100 years, as described in Section D.4.3.4.1 (Benzo(a)pyrene).

D.4.5 Deposition-Soil-Plant Toxicity Exposure Pathway

D.4.5.1 Assumptions. The assumptions pertinent to this pathway have been stated in Table D-2.

D.4.5.2 Calculation Method. For this pathway, exposure is equivalent to the deposition rates or soil concentrations of contaminants deposited from municipal waste combustor emissions.

D.4.5.3 Example Calculations. No calculations are required for this pathway, except in some cases to convert CD to LC.

D.4.6 Exposure Pathways for Toxicity to Soil Biota and Their Predators

This section deals with two pathways: toxicity to soil biota (deposition-soil-soil biota toxicity) and toxicity to predators of soil biota (deposition-soil-soil biota-predator toxicity). The term "soil biota" refers to a broad range of organisms including microorganisms and various invertebrates living in or on the soil. Their predators similarly include a variety of organisms. The availability of data determines what species are considered.

D.4.6.1 Deposition-Soil-Soil Biota Toxicity Exposure Pathway. For this pathway exposure is equivalent to the deposition rates or soil concentrations of contaminants deposited from municipal waste combustors.

D.4.6.1.1 Example calculations. This pathway does not require calculations, except conversion of CD to LC.

D.4.6.2 Deposition-Soil-Soil Biota-Predator Toxicity Exposure Pathway.

D.4.6.2.1 Assumptions. In addition to many of the assumptions listed in Table D-7, some additional assumptions relevant to this pathway are stated Table D-8.

D.4.6.2.2 Calculation method. Calculations of criteria for this pathway may take either of two forms, depending on the type of data available concerning contaminant uptake by soil biota. If uptake response (increase in concentration) in soil biota, UB, can be expressed in terms of a contaminant deposition rate, the predator intake is calculated as follows:

TABLE D-7. ASSUMPTIONS FOR ECOLOGICAL EFFECTS FOR TERRESTRIAL FOOD CHAIN

Functional Area	Assumptions	Ramifications/Limitations
Toxicity thresholds for nonhuman organisms	All inhibitory effects should be considered adverse.	Some "inhibitory" changes might not significantly affect the individual survival or population dynamics. Conversely, some ecologically important effects might not be observed in toxicity tests.
	The geometric mean of exposure levels bracketing the appearance of an adverse effect should be used as the threshold.	The true threshold may lie at any point between these two levels, and may be over- or underpredicted by this method.
	The form of a contaminant used in a study should not be considered equally bioavailable and toxic as form in soil, unless suitable data using soil are not available.	Availability of chemicals in soil or deposited on soil may differ; particularly, it may be lower, so that toxicity is overpredicted.

TABLE D-8. ASSUMPTIONS FOR DEPOSITION-SOIL-SOIL BIOTA-PREDATOR TOXICITY EXPOSURE PATHWAY

Functional Area	Assumptions	Ramifications/Limitations
Contaminant uptake by soil biota	Response in tissues of soil biota can be represented by a linear function.	Probably oversimplifies a more complex relationship.
Use of available data to protect a variety of species	It is assumed that use of the highest available response slope in soil biota and the lowest available dietary threshold in predators will result in protection of untested species.	This conservative assumption could be overprotective of some species; the extent to which it underprotects others is unknown. A "match" of data for a consumed organism and its predator usually is not possible.

$$PFC = CD \times UB \qquad \text{Equation (D-10)}$$

where:

PFC = predator feed concentration (ug/g DW)
CD = cumulative soil deposition of pollutant (kg/ha)
UB = uptake response slope in soil biota (ug/g $[kg/hg]^{-1}$

If the chemical is not subject to degradation or loss in soil, PFC is a cumulative concentration. If soil biota response is measured in terms of a soil concentration, the following equation is used:

$$PFC = LC_T \times UB \qquad \text{Equation (D-11)}$$

where:

LC_T = maximal soil concentration of pollutant at time, T (ug/g DW)
UB = uptake response slope in soil biota (ug/g $[ug/g]^{-1}$)

LC represents cumulative soil concentration (above background) due to emissions from municipal waste combustors.

D.4.6.2.3 Input parameter requirements.

Uptake response slope in soil biota (UB)--UB may take any of several forms, depending on the characteristics of the chemical and the data available. UB is derived by linear regression of tissue concentration by either contaminant deposition rate or soil concentration. Two or more points are needed to derive the slope for inorganics, while for organic compounds a single data pair can be used to derive a bioconcentration factor for plants and animal tissues.

The highest available uptake response slope will be used to estimate the level of dietary contamination to which predators of soil biota would be subject. This fact is important when evaluating data for earthworms. Some studies distinguish between contaminant physiologically absorbed and that which is due to gut contents or soil contamination of the sample. There is no need to distinguish between absorbed and unabsorbed contaminants, since a predator would ingest the gut contents as well as the rest of the organism. Whenever possible, analyses used should be based on the whole organism.

D.4.6.2.4 Example calculations.

Cadmium--The predator feed concentration (PFC) for cadmium is calculated using Equation (D-11):

$$PFC = CD \times UB$$

where:

CD = cumulative soil deposition of pollutant (kg/ha)
UB = uptake response slope in soil biota ($ug/g[kg/ha]^{-1}$)

CD is 3.26 and 10.88 kg/ha for 30 and 100 years, respectively, for cadmium, and the UB is 13.7 ug Cd/g $(kg/ha)^{-1}$.[31] The PFC for cadmium for 30 years is,

$$PFC = 3.26 \text{ kg/ha} \times 13.7 \text{ ug/g } (\text{kg/ha})^{-1} = 44.66 \text{ ug/g}$$

For 100 years,

$$PFC = 10.88 \text{ kg/ha} \times 13.7 \text{ ug/g } (\text{kg/ha})^{-1} = 149.06 \text{ ug/g}$$

Benzo(a)pyrene--Lack of data availability for UB of B(a)P prohibits calculation of a PFC.

D.5 REFERENCES FOR APPENDIX D

1. O'Flaherty, E. J. Toxicants and Drugs: Kinetics and Dynamics. John Wiley and Sons. New York, New York. 1981.

2. U. S. Environmental Protection Agency. Effects of Sewage Sludge on the Cadmium and Zinc Content of Crops. Council for Agricultural Science and Technology. EPA Report No. 600/8-81-003. 1980.

3. Ryan, J. A., H. R. Pahren, and J. B. Lucas. Controlling Cadmium in the Human Food Chain - A Review and Rationale Based on Health Effects. Environ. Res. 28:251-302. 1982.

4. Logan, T. J., and R. L. Chaney. Utilization of Municipal Wastewater and Sludge on Land - Metals. Proceedings of the Workshop on Utilization of Municipal Wastewater and Sludge on Land. University of California at Riverside, Riverside, CA. pp. 253-323. 1983.

5. Reference 3.

6. Page, A. L., T. J. Logan, and L. E. Summers. Draft Report of the Workshop on Effects of Sewage Sludge Quality and Soil Properties on Plant Uptake of Sludge - Applied Trace Contaminants. Las Vegas, Nevada. 1986.

7. Reference 3.

8. Pennington, J. Revision of the Total Diet Study Food List and Diets. J. Amer. Diet Assn. 82:166-173. 1983.

9. U. S. Environmental Protection Agency. Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge. Draft Final Report. 1986.

10. Food and Drug Administration. Documentation of the Revised Total Diet Study Food Lists and Diets. NTIS Publication No. 82-192154. 1981.

11. U. S. Department of Agriculture. Composition of Foods. Agricultural Handbook No. 8. 1975.

12. U. S. Department of Agriculture. Household Food Consumption Survey, 1965-1966. Report 12. Food Consumption of Households in the United States - Seasons and Year, 1965-1966. 1966.

13. Reference 8.

14. Dowdy, R. H., and W. E. Larson. The Availability of Sludge-Borne Metals to Various Vegetable Crops. J. Environ. Qual. 4:278-282. 1975.

15. Reference 9.

16. Reference 13.

17. Connor, M. S. Monitoring Sludge - Amended Agricultural Soils. Biocycle 25:47-51. 1984.

18. Reference 9.

19. Reference 17.

20. Bosert, D. D., and R. Bartha. Structure - Biodegradability Relationships of Polycyclic Aromatic Hydrocarbons in Soil. Bulletin of Environmental Contamination and Toxicology 37:490-495. 1986.

21. National Resource Council. Mitigative Techniques and Analysis of Generic Site Conditions for Groundwater Contamination Associated with Severe Accidents. Battelle Memorial Institute. 1984.

22. U. S. Environmental Protection Agency. Air Quality Criteria for Lead. EPA Report No. 600/8-83-0288.

23. Lepow, M. L., M. Gillette, S. Markowitz, R. Robino, and J. Kapish. Investigations into Sources of Lead in the Environmental of Urban Children. Environ. Res. 10:415-426. 1975.

24. Reference 22.

25. Binder, S., D. Sokal, and D. Maughan. Estimating the Amount of Soil Ingested by Young Children through Trace Elements. Draft Report. Centers for Disease Control, Center for Environmental Health, Special Studies Branch. 1985.

26. Reference 9.

27. Thornton, D., and P. Abrams. Soil Ingestion: A Major Pathway of Heavy Metals into Livestock Grazing Contaminated Land. Sci. Total Environ. 28:287-294. 1983.

28. Telford, J. N., M. L. Thonney, and D. E. Hogue. Toxicological Studies in Growing Sheep Fed Silage Corn Cultured on Municipal Sludge - Amended Acid Subsoil. J. of Toxicol. Environ. Health 10:73-85. 1982.

29. U. S. Environmental Protection Agency. Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: Cadmium. 1985.

30. U. S. Environmental Protection Agency. Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: Benzo(a)pyrene. 1985.

31. Reference 30.

APPENDIX E
SURFACE RUNOFF MODEL

APPENDIX E
SURFACE RUNOFF MODEL

E.1 INTRODUCTION

Contaminants associated with particulates emitted by municipal waste combustors are subject to deposition on surfaces downwind from the municipal waste combustor at rates determined by meteorology, terrain and particle physics. This fallout is subsequently subject to dissolution and/or suspension in runoff after precipitation events. Runoff moves over the surface of the earth to a surface water body where it mixes with other waters. As a consequence, humans utilizing water or eating fish from the surface water body or aquatic life living therein may be exposed to runoff transported contaminants.

The methodology derived to calculate exposure from the surface runoff pathway was originally developed to evaluate impacts from the application of municipal wastewater sludge to land. A detailed discussion is available in the document entitled "Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge."[1] The methodology is formulated in three successive tiers, which begin with simple but very conservative estimates, and proceed to more detailed analyses if the first tiers predict unacceptable exposures. Both acute events and chronic exposure are evaluated using standard approaches to calculate runoff volume and associated erosion potential.

The remainder of this appendix describes the assumptions, calculation steps, and input requirements of the Surface Runoff Model. Example calculations are also presented.

E.2 ASSUMPTIONS

A number of assumptions were required to formulate the risk-based methodology both with respect to runoff generation and subsequent mixing in the receiving water. The key assumptions are provided in Table E-1 with a

TABLE E-1. RFACE RUNOFF METHODOLOGY ASSUMPTIONS

Functional Area	Assumption	Ramifications
Long-Term Concentrations:		
Tier 1	All contaminant deposited on an annual basis is transported to the receiving water in a dissolved form.	Provides an extremely conservative estimate since no losses are considered.
Tier 2/3 General	Loadings to the receiving water can be described as a function of solids loading.	Mechanistically inappropriate for contaminants with low partition coefficients.
Source area	Facility operates over a sufficient period for surface soil levels to reach equilibrium where annual losses equal annual inputs.	Overpredicts contaminant loading by ignoring loss mechanisms other than those used in formulation, namely, runoff and infiltration.
	No settling of particles in the deposition zone, gross erosion reaches the edge of field.	Maximizes contaminant loss.
Event concentrations:		
Tier 1	All contaminant emitted in a year is lost in a single runoff event.	Provides extremely conservative estimate with no provision for losses or incomplete mobilization.
Tier 2/3 Source area	No kinetically limited release of contaminant from residual/soil mixture; i.e., total contaminant concentration is fully equilibrated into adsorbed and dissolved phases.	Maximizes contaminant concentration available for runoff in dissolved phase, thereby maximizing loadings.
Stream	Stream flow is unchanged by the storm unless arterial velocity data are available from the hydrograph.	Overestimates stream concentration, since the storm will increase stream flow.

discussion of their impact on the methodology. Because the science is not exact, the assumptions are mostly conservative (i.e., overpredict contaminant concentrations). In some instances, however, the nature of the effect of a given assumption may vary with site-specific considerations.

E.3 CALCULATION STEPS

The methodology addresses both long- and short-term exposures as illustrated in Figure E-1. In Tier 1, it is assumed that all contaminant emitted in a given year is transported to the receiving water in that year. The total mass flux of contaminant is distributed among the watersheds in which the fallout is deposited. The downsteam boundary flow is then used to calculate the resulting concentration in the receiving water:

$$Ci_x = [(Fa)\ (WA_x)]/Vf_x \qquad \text{Equation (E-1)}$$

where:

Ci_x = estimated receiving water concentration of contaminant for watershed x (M/L^3)

Fa = annual mass of contaminant in fallout/unit area (M/L^2-T)

WA_x = area of watershed receiving fallout (L^2)

Vf_x = total volume of flow of watershed outlet during the period of observation; i.e., 1 year for chronic exposure (L^3/T)

For chronic exposure, total annual flow, Vf_x, is applied, while for acute exposure the total flow over the duration of the storm event, VS_x, is substituted (e.g., $VS_x = Vf_x/365$), if the 24-hour rainfall event is evaluated. (Ideally, VS_x is the actual hydrograph velocity for the event if it is known.) This very conservative calculation accounts for no losses during transport and, therefore, overpredicts contaminant levels. If these overpredictions do not result in excessive concentrations, no further analysis is required. If the concentration predictions are high, a more detailed Tier 2 or 3 analysis is required to determine probable receiving water concentrations.

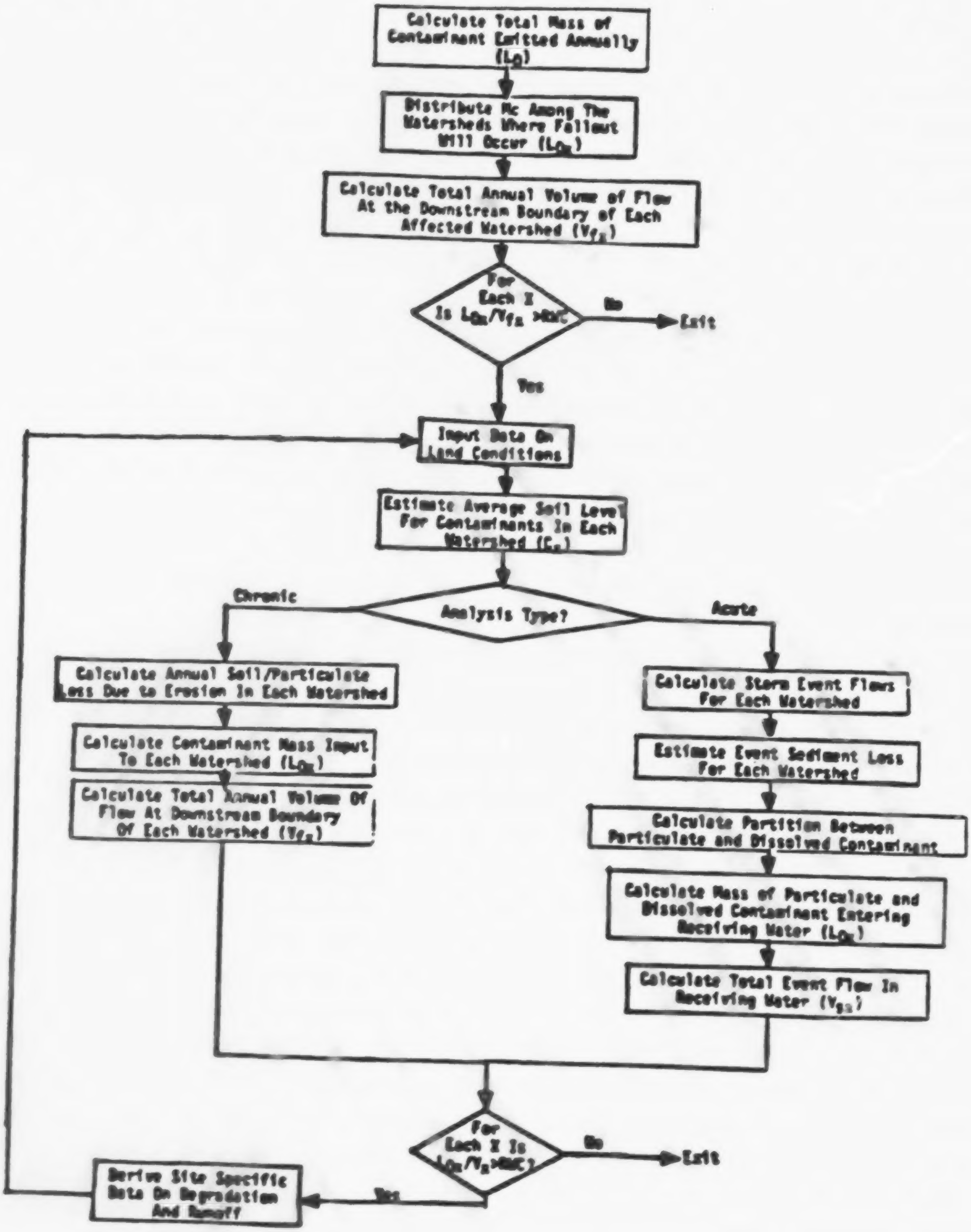
Calculate Total Mass of Contaminant Emitted Annually (L_0)
Distribute Mc Among The Watersheds Where Fallout Will Occur (L_{0x})
Calculate Total Annual Volume of Flow At the Downstream Boundary of Each Affected Watershed (V_{fx})
For Each X Is L_{0x}/V_{fx} >RMC
No
Exit
Yes
Input Data On Land Conditions
Estimate Average Soil Level For Contaminants In Each Watershed (C_x)
Chronic
Analysis Type?
Acute
Calculate Annual Soil/Particulate Loss Due to Erosion In Each Watershed
Calculate Contaminant Mass Input To Each Watershed (L_{0x})
Calculate Total Annual Volume Of Flow At Downstream Boundary Of Each Watershed (V_{fx})
Calculate Storm Event Flows For Each Watershed
Estimate Event Sediment Loss For Each Watershed
Calculate Partition Between Particulate and Dissolved Contaminant
Calculate Mass of Particulate and Dissolved Contaminant Entering Receiving Water (L_{0x})
Calculate Total Event Flow In Receiving Water (V_{sx})
For Each X Is L_{0x}/V_x >RMC?
No
Exit
Yes
Derive Site Specific Data On Degradation And Runoff

Figure E-1. Surface Runoff Pathway Methodology

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Tier 2 and 3 calculations are identical. They differ only in the origin of input values. Tier 3 is based on site-specific data from empirical observations for such parameters as degradation rate and partition coefficient. For the long-term (chronic) exposure analysis, receiving water contaminant inputs are calculated from estimates of the soil contaminant concentration and the bulk soil transport to the receiving water.

In order to estimate soil contaminant concentration, it is assumed that the municipal waste combustors are in operation long enough for a steady-state concentration to be reached. By definition, these conditions will prevail when soil levels are high enough for the sum of zeroth and first-order losses to equal the annual addition of contaminants in fallout. Maximum soil contaminant mass per area during the period is calculated as follows:

$$Mm = \frac{k_2(1-3^{-k_1 t})}{k_1} \qquad \text{Equation (E-2)}$$

where:

Mm = maximum contaminant mass per area of soil (M/L^2)

k_2 = annual fallout rate for contaminant less any zeroth order losses (typically none) (M/L^2-T)

k_1 = first-order loss rate includes infiltration losses, which are controlled by partitioning, degradation and erosion losses (T^{-2})

t = time span for analysis, typically the life of the combustor and its replacements

The first-order loss mechanisms can be calculated as follows:

1) Infiltration (if equilibrium between soil and water is assumed)

$$K_{1I} = Rc/(B\ d\ Kd) \qquad \text{Equation (E-3)}$$

where:

k_{1I} = first-order loss rate coefficient for infiltration (T^{-1})

Rc = recharge (L/T)

B = the bulk density (M/L^3)
d = depth of incorporation (L)
Kd = the distribution coefficient (L^3/M)

Once again, if infiltrate concentrations are thought to be solubility limited, a zeroth order rate is more appropriate.

2) Surface runoff

$$k_{1R} = X_e/(B\ d) \qquad \text{Equation (E-4)}$$

where:

k_{1R} = first-order loss rate coefficient for surface runoff losses (T^{-1})
X_e = sediment loss rate (M/L^2-T)
B = bulk density (M/L^3)
d = depth of incorporation (L).

3) Degradation

$$k_{1D} = \ln 2/t_{1/2} \qquad \text{Equation (E-5)}$$

where:

k_{1D} = first-order loss rate for degradation (T^{-1})
$t_{1/2}$ = half-life due to degradation (T).

In general, volatilization would also represent a first-order loss mechanism. For particulate fallout, however, it is assumed that volatile species will not be present in the solids settling from the atmosphere. Even if vapor pressures are measurable, the adsorption phenomena will reduce their significance. Also, when field measurements are available for deriving degradation rates, they may reflect all first-order losses (k_1) and not just degradation (LC_{LD}). Therefore, care must be taken in selecting input values.

If tilling is prevalent in the watershed, concentrations are reduced by dividing by d (depth of tilling in cm) to reflect that runoff only affects the top centimeter of soil. When no tilling is practiced, d is set at 1.0 cm.

Sediment losses to the receiving water are computed using the Universal Soil Loss Equation (USLE):[2]

$$X_e = R\ K\ (LS)\ C\ P \qquad \text{Equation (E-6)}$$

where:

X_e = sediment loss rate (M/L^2-T)
R = "erosivity" factor (T^{-1})
K = "erodability" factor (M/L^2-T-unit 'R')
LS = "topographic or slope length" factor (dimensionless)
C = "cover management" factor (dimensionless)
P = "supporting practice" factor (dimensionless)

Guidelines for selection of input factor values for the USLE in each watershed are provided by Wischmeier and Smith as well as the detailed runoff methodology discussion.[3,4]

The total annual flux of sediment to a receiving water is the product of the unit area sediment loss, X_e, and the area of the watershed, WA_x. The concentration of the contaminant in the receiving water (Ci_x) is determined as the ratio of the total contaminant flux and the annual volume of flow at the downstream watershed boundary, Vf_x, or:

$$Ci_x = X_e\ WA_x\ Mm/(d\ Vf_x\ B) \qquad \text{Equation (E-7)}$$

where:

X_e = sediment loss rate (M/L^2-T)
WA_x = area of watershed receiving fallout (L^2)
Mm = maximum contaminant mass per area of soil (M/L^2)
d = depth of tilling (L)
Vf_x = total volume of flow at watershed outlet during the period of observation (L^3/T)
B = bulk density of soil (M/L^3)

The point for calculating the annual flow should be the furthest downstream point where the contaminated watershed feeds the receiving water. For lakes or estuaries, the outlet flow is applied as Vf_x. Once again, these calculations are made for each affected watershed or subwatershed unit.

For acute exposures, the methodology focuses on a specific storm event. To estimate average contaminant mass per area of soil, it is once again assumed that soil levels will increase until balanced by losses. The maximum level (Mm) at any time (t) after the combustion is initiated can be estimated as:

$$Mm = \frac{k_2(1-e^{-k_1 t})}{k_1} \qquad \text{Equation (E-8)}$$

where the terms are the same as defined in Equation (E-2). Once again, if tilling is practiced to a depth of d cm, the contaminant mass per unit volume of soil (Mm', in M/L^3) is:

$$Mm' = Mm/d = \frac{k_2(1-e^{-k_1 t})}{d\ k_1}$$

Sediment loss to the receiving water is calculated from the Modified Universal Loss Equation (MUSLE).[5,6] In this approach, it is necessary first to select a watershed retention factor (S) from the Soil Conservation Service (SCS) runoff curve number (CN) according to:

$$S = 2.54[(1000/CN)-10] \qquad \text{Equation (E-9)}$$

where:

S = watershed retention factor (cm)

Note that the units for S are centimeters and that S should be converted to the proper length units for the remainder of the calculations. Using length units of centimeters for D_R, R_t, M_t and S is preferred. Next, a runoff depth value is estimated as:

$$D_R = \frac{(R_t + M_t - 0.2S)^2}{(R_t + M_t + 0.8S)} \qquad \text{Equation (E-10)}$$

where:

D_R = depth of runoff in the watershed (L)
R_t = depth of total rainfall for the storm event (L)
M_t = depth of snowmelt during the storm event (L)
S = the watershed retention parameter (L)

The storm runoff volume is calculated from the runoff depth according to:

$$Q = (WA_x)D_R \qquad \text{Equation (E-11)}$$

where:

WA_x = area of the watershed receiving fallout (L^2)
Q = volume of runoff (L^3)
D_R = depth of runoff from the storm event (L)

A trapezoidal hydrograph is assumed so that the peak runoff rate can be calculated as:

$$q_p = \frac{(WA_x)\ D_R\ R_t}{T_r(R_t - 0.2S)} \qquad \text{Equation (E-12)}$$

where:

q_p = peak runoff rate (L^3/T)
WA_x = area of the watershed (L^2)
R_t = depth of rainfall in the storm event (L)
D_R = depth of runoff from the storm event (L)
T_r = duration of the storm event (T)
S = water retention parameter (L)

Sediment losses from the storm event are estimated with the MUSLE according to:

$$X_s = 11.8(Q\ q_p)^{0.56} K\ (LS)\ C\ P \qquad \text{Equation (E-13)}$$

where:

X_s = sediment loss from a single storm (metric tons)
Q = volume of runoff (m^3)
q_p = peak runoff (m^3/sec)
K = "erodability" factor (tons/acre-year-unit "R")
LS = "topographic or slope/length" factor (dimensionless)
C = "cover management" factor (dimensionless)
P = "supporting practice" factor (dimensionless)

Again, selection of K, LS, C and P is discussed elsewhere.[7,8] This equation is an empirical relationship and the units must be consistent with those shown above.

If the risk evaluation is to be based on total contaminant exposure, receiving water concentrations are calculated as in Equation (E-7):

$$Ci_x = X_s \; WA_x \; Mm'/VS_x \; B \qquad \text{Equation (E-14)}$$

using the maximum soil contaminant level Mm' in place of the average soil contaminant level MC. The value for VS_x (used instead of Vf_x) would be the total volume of flow of the receiving water during the storm event rather than a 1-year period. If criteria for risk differentiate between dissolved and particulate contaminant, then the total mass of contaminant Mm' must be partitioned between the adsorbed portion, Aa, and the dissolved portion, Da. These are derived as:

$$Aa = [1/(1+(\theta/Kd \; B))]Mm' \qquad \text{Equation (E-15)}$$

and

$$Da = [1/(1+(Kd \; B/\theta))]Mm' \qquad \text{Equation (E-16)}$$

where:

Aa = adsorbed contaminant mass in top cm of soil (M/L^2-L)
Da = dissolved contaminant mass in top cm of soil (M/L^2-L)
θ = available volumetric water capacity of the top cm of soil difference between wilting point and field capacity) (dimensionless)
Kd = distribution coefficient for contaminant in soil water system (L^3/M)
B = bulk density of soil (M/L^3)
Mm' = maximum level of contaminant in top centimeter of watershed soil (M/L^2-L)

The contaminant losses by each route are defined as:

$$F_{xt} = [X_s/(WA_x \; B)]Aa \qquad \text{Equation (E-17)}$$

and

$$P_{qt} = [D_R/(R_t + M_t)]Da \qquad \text{Equation (E-18)}$$

where:

P_{xt} = loss of contaminant in adsorbed form (M/L^2)
P_{qt} = loss of contaminant in dissolved form (M/L^2)

Other terms are as defined in Equations (E-10), (E-14), (E-15) and (E-16). For these cases, the receiving water concentrations would be derived from:

and

$$Ci_x = P_{xt}\ WA_x/VS_x \qquad \text{Equation (E-19)}$$

$$Ci_x = P_{qt}\ WA_x/VS_x \qquad \text{Equation (E-20)}$$

where:

Ci_x = concentration in receiving water (M/L^3)
P_{qt} and P_{xt} = loss rates from Equations (E-17) and (E-18) (M/L^2)
WA_x = area of watershed (L^2)
VS_x = total volumetric flow of receiving water during the storm event (L^3)

E.4 REQUIRED INPUTS

For a Tier 1 analysis, the only input parameters required are the annual mass of contaminant fallout on each watershed or subwatershed unit and the annual flow of the receiving water.

The equations for Tier 2 and Tier 3 analysis are the same; therefore, they have the same input parameters. The difference is that many of the parameters used in a Tier 2 analysis would be obtained from the literature, whereas Tier 3 requires all site-specific input. All the input parameters required for the runoff pathway analysis are shown in Table E-2. The only input required for the receiving water analysis is the stream flow or the flow into a lake or estuary.

E.5 EXAMPLE CALCULATION

The methodology presented above can best be illustrated with example calculations. Two site-specific examples are provided below, one for a long-term average case (chronic) and one for an event-loading (acute case). For both cases, Tier 1 and Tier 2/3 analyses are made for two contaminants, benzo(a)pyrene and cadmium. Contaminant deposition patterns were modeled for emissions representative of a municipal waste combustor with an electrostatic precipitator.

The annual mass of contaminant fallout (Fa) was calculated by averaging deposition estimates for a given area over a 5-year period. Deposition rates were estimated at points on rays spaced every 22.5° at ranges varying

TABLE E-2. INPUT PARAMETERS FOR THE RUNOFF PATHWAY METHODOLOGY

Symbol	Function
f_{oc}	Soil organic content (dimensionless)
WA_x	Area of land in a given watershed on which fallout will be deposited (km²)
Fa	Annual fallout rate/watershed unit (dry kg/ha-year)
t	Time period over which facility and its replacement will operate (years)
R_c	Recharge rate (m/year)
B	Bulk density of the soil (g/cm³)
d	Depth of soil incorporation*
M_t	Total event snow melt (cm)
R_t	Total storm rainfall depth (cm)
T_r	Duration of storm event (hours)
Kd	Distribution coefficient (cm³/g)
Vf_x	Total volume streamflow for the receiving water (ℓ/year)
VS_x	Event streamflow volume (ℓ/event)
R, K, LS, C and P	Input variables for the Universal Soil Loss Equation (USLE)
θ	Soil porosity (dimensionless)
$t_{1/2}$	Contaminant half-life (1/years)

*d is used even if not tilled

from 200 to 50,000 m from the municipal waste combustor. Hence, the sample points form concentric rings with the combustor at their center. The average annual mass of contaminant fallout was obtained by integrating the measured data over the watershed area (1 km^2) being studied. The double integral:

$$1/\pi R^2 \iint f(r,\theta)\ r\ dr\ d\theta$$

where:

$$f(r,\theta) = \frac{f(500,\theta) - f(200,\theta)}{300}\ r + 5/3f(200,\theta)2/3f(500,\theta)$$

was evaluated for all data points on the 200 and 500 m rings (the area within the 500 m ring is roughly the size of 1 km^2 watershed). Integrating r's from 0 to 500 m and θ's from 0 to 2π, the resulting average benzo(a)-pyrene and cadmium fallout was 0.0020779 kg/ha-year and 0.039436 kg/ha-year, respectively.

It is assumed that the particulate emitted by the municipal waste combustor is spread over various watersheds; however, for the purpose of illustration, the calculations are made for only a single watershed of an area roughly 1 km^2 that contains the MWC facility at its center.

All input parameters for the example calculations are shown in Table E-3. Data values are for illustrative purposes only, but are, for the most part, representative of a real site.

E.5.1 Tier 1

E.5.1.1 Long-Term Average Loading. For a Tier 1 analysis, calculate the maximum possible receiving water concentration using Equation (E-1). For benzo(a)pyrene:

$$\begin{aligned} Ci_x &= [10^8(Fa_x \times WA_x)]Vf_x \\ &= 10^8(0.0020779\ \text{kg/ha} \times 1.0\ \text{km}^2)/3.15 \times 10^{10}\ \text{l/year} \\ &= 6.5965 \times 10^{-6}\ \text{mg/l} \end{aligned}$$

For cadmium:

$$\begin{aligned} Ci_x &= [10^8(Fa_x \times WA_x)]Vf_x \\ &= 10^8(0.0039436\ \text{kg/ha} \times 1.0\ \text{km}^2)/3.15 \times 10^{10}\ \text{l/year} \\ &= 0.0001252\ \text{mg/l} \end{aligned}$$

TABLE E-3. INPUT PARAMETERS FOR THE EXAMPLE CALCULATIONS

Location - western Florida
Soil Type - sandy clay loam
Organic matter content (f_{oc}) - 4%
Land use - agricultural, orchards

WA - watershed area, 1 km² (100 ha)
Fa - annual mass fallout/unit area:
    benzo(a)pyrene - 0.0020779 kg/ha-year
    cadmium - 0.039436 kg/ha-year
t - elapsed time, 30 years
Rc - recharge rate, 0.25 m/year
Vf_x - annual streamflow volume, 3.15×10^{10} ℓ/year
B - soil bulk density, 1.5 g/cm³
d - incorporation depth, 1 cm

k_0 - zeroth order loss rate, 0.0
k_1 - first-order loss rate
    benzo(a)pyrene - degradation + infiltration and surface runoff, 0.278 year^{-1}
    cadmium - infiltration + surface runoff, 0.168 year^{-1}
k_2 - $Fa - k_0 = Fa$

USLE Input Parameters:

R - 400 year^{-1}
K - 0.21 tons/acre/year/unit R=0.21 tons/acre
LS - 0.179
C - 0.5
P - 1.0
CN - 78

R_t - total event storm rainfall, 5 cm
M_t - total event snow melt, 0 cm
T_r - storm duration, 6 hours
VS_x - event/storm streamflow volume, 8.63×10^7 ℓ/event
Kd - partition coefficient:
    benzo(a)pyrene - 3000 cm³/g
    cadmium - 300 cm³/g
θ - porosity, 0.2

E.5.1.2 Event Loading. For event loading, VS_x is defined as the total volume of flow during the selected storm event rather than the 1-year flow volume (Vf_x) used for the long-term analysis. For this illustration, a 24-hour event is assumed. Therefore, the flow volume would be:

$$8.63 \times 10^7 \text{ l } (3.15 \times 10^{10} \text{ l/year} \div 365 \text{ day/year})$$

The concentration of benzo(a)pyrene in the receiving water would be:

$$Ci_x = [10^8 (Fa_x)(WA_x)]/VS_x$$

$$= 10^8(0.0020779 \text{ kg/ha})(1 \text{ km}^2)/8.63 \times 10^7 \text{ l/event})$$

$$= 0.002408 \text{ mg/l}$$

The concentration of cadmium in the receiving water would be:

$$Ci_x = [10^8(Fa_x)(WA_x)]/VS_x$$

$$= 10^8(0.0039436 \text{ kg/ha})(1 \text{ km}^2)/8.63 \times 10^7 \text{ l/event})$$

$$= 0.04570 \text{ mg/l}$$

E.5.2 Tier 2/3

E.5.2.1 Long-Term Loading. To estimate soil contaminant concentration it is first necessary to estimate rate constants for contaminant loss from soils. For a degradable contaminant, k_1 is the combined first-order loss rate constant for degradation, infiltration and surface runoff. For a nondegradable contaminant, k_1 is based on the loss due to infiltration and surface runoff only. For cadmium (nondegradable), k_1 was calculated as follows:

1) Infiltration

$$k_{1I} = Rc/(B\ d\ Kd)$$
$$= 0.25\ \text{m/year}/(1.5\ \text{g/cm}^3)(1\ \text{cm})(300\ \text{cm}^3/\text{g})$$
$$= 0.0556/\text{year}$$

2) Surface Runoff

$$k_{1R} = X_e/(B\ d)$$
$$= 10^{-4}\ [1688\ \text{mt/km}^2\text{-year}/(1.5\ \text{g/cm}^3)(1\ \text{cm})]$$
$$= 0.1125/\text{year}$$

$$k_1 = k_{1I} + k_{1R}$$
$$= 0.0556/\text{year} + 0.1125/\text{year}$$
$$= 0.168/\text{year}$$

For benzo(a)pyrene (degradable) a biodegradation rate in soil of 0.16/year was estimated from a recent study,[9] and k_1 was calculated as follows:

1) Infiltration

$$k_{1I} = Rc/(B\ d\ Kd)$$
$$= 0.25\ \text{m/year}/(1.5\ \text{g/cm}^3)(1\ \text{cm})(3000\ \text{cm}^3/\text{g})$$
$$= 0.0056/\text{year}$$

2) Surface Runoff

Same as 2) above where $k_{1R} = 0.1125/\text{year}$

3) Degradation

$$k_{1D} = 0.16/\text{year}$$

$$k_1 = k_{1I} + k_{1R} + k_{1D}$$

$$= 0.0056/\text{year} + 0.1125/\text{year} + 0.16/\text{year}$$

$$= 0.278/\text{year}$$

Following estimation of degradation rates, the soil contaminant concentration may be estimated.

For a degradable contaminant, such as benzo(a)pyrene, the soil contaminant concentration can be estimated with Equation (E-2) as:

$$
\begin{aligned}
Mm &= \frac{k_2}{k_1}(1-e^{-k_1 t}) \\
&= \frac{0.0020779 \text{ kg/ha-year}}{0.278 \text{ year}^{-1}} [(1-e^{-(0.278 \text{ year}^{-1}) (30 \text{ years})}] \\
&= 0.007473 \text{ kg/ha}
\end{aligned}
$$

For a nondegradable contaminant, cadmium, the maximum soil contaminant concentration can be estimated with Equation (E-2) as:

$$
\begin{aligned}
Mm &= \frac{k_2}{k_1}(1-e^{-k_1 t}) \\
&= \frac{0.039436 \text{ kg/ha-year}}{0.278 \text{ year}^{-1}} [(1-e^{-(0.168 \text{ year}^{-1}) (30 \text{ years})}] \\
&= 0.23322 \text{ kg/ha}
\end{aligned}
$$

Bulk soil losses to the receiving water are calculated using Equation (E-6) as follows:

$$
\begin{aligned}
X_e &= 224.64(R)(K)(LS)(C)(P) \\
&= (224.64)(400/\text{year})(0.21 \text{ tons/acre/year})(0.179)(0.5)(1.0) \\
&= 1688 \text{ mt/km}^2\text{-year}
\end{aligned}
$$

The total annual flux of sediment to a receiving water is the product of the unit area sediment loss (X_e) and the area of the watershed (1 km^2). The contaminant concentration of benzo(a)pyrene in the receiving water is calculated with Equation (E-2) as:

$$Ci_x = \frac{(10^4)(X_e)(WA_x)(Mm)}{(d)(Vf_x)(B)}$$

$$= \frac{(10^4)(1688\ \text{mt/km}^2\text{-year})(1\ \text{km}^2)(0.007473\ \text{kg/ha})}{(1\ \text{cm})(3.15 \times 10^{10}\ \text{l/year})(1.5\ \text{g/cm}^3)}$$

$$= 2.669 \times 10^{-6}\ \text{mg/l}$$

For cadmium:

$$Ci_x = \frac{(10^4)(X_e)(WA_x)(Mm)}{(d)(Vf_x)(B)}$$

$$= \frac{(10^4)(1688\ \text{mt/km}^2\text{-year})(1\ \text{km}^2)(0.23322\ \text{kg/ha})}{(1\ \text{cm})(3.15 \times 10^{10}\ \text{l/year})(1.5\ \text{g/cm}^3)}$$

$$= 8.332 \times 10^{-6}\ \text{mg/l}$$

E.5.2.2 Event Loading. For the degradable contaminant, benzo(a)-pyrene, the maximum contaminant level in the soil at any time can be calculated as above using Equation E-8 as:

$$Mm = \frac{k_2(1-e^{-k_1 t})}{k_1}$$

$$= \frac{0.0020779\ \text{kg/ha-year}}{0.278\ \text{year}^{-1}} \left[(1-e^{-(0.278\ \text{year}^{-1})\ (30\ \text{years})}\right]$$

$$= 7.473 \times 10^{-3}\ \text{kg/ha}$$

If a soil incorporation depth of 1 cm is assumed, then:

$$Mm' = Mm/(1\ \text{cm})$$

$$Mm' = 7.473 \times 10^{-3}\ \text{kg/ha}\bullet\text{cm}$$

For the nondegradable contaminant cadmium:

$$Mm = \frac{k_2(1-3^{-k_1 t})}{k_1}$$

$$= \frac{0.039436 \text{ kg/ha-year}}{0.168 \text{ year}^{-1}} [(1-e^{-(0.168 \text{ year}^{-1})\ (30 \text{ years})}]$$

$$= 0.2332 \text{ kg/ha}$$

$$Mm' = 0.2332 \text{ kg/ha} \bullet \text{cm}$$

where CN corresponds to hydrologic soil Group B, moderately low runoff potential, good hydrologic conditions and straight row crops.

The runoff depth can be calculated using Equation (E-10) as:

$$D_R = \frac{(R_t + M_t - 0.2S)^2}{(R_t + M_t + 0.8S)}$$

$$= \frac{[5 \text{ cm} + 0 - (0.2)(7.16 \text{ cm})]^2}{5 \text{ cm} + 0 + (0.8)(7.16 \text{ cm})}$$

$$= 1.19 \text{ cm}$$

Calculate the storm runoff volume from Equation (E-11) as:

$$Q = 10^4(WA_x)D_R$$

$$= (10^4)(1 \text{ km}^2)(1.19 \text{ cm})$$

$$= 1.19 \times 10^4 \text{ m}^3$$

Calculate the peak runoff rate from Equation (E-12) as:

$$q_p = \frac{(2.78)(WA_x)(R_t)(D_R)}{{}_rT\ (R - 0.2S)}$$

$$= \frac{(2.78)(1\ km^2)(5\ cm)(1.19\ cm)}{(6\ hours)\ [5\ cm - 0.2\ (7.16\ cm)]}$$

$$= 0.77\ m^3/sec$$

The sediment losses from the storm event are calculated from the MUSLE Equation (E-13) as:

$$X_s = 1.8\ (Q\ q_p)^{0.56}\ (K)\ (LS)\ (C)\ (P)$$

$$= 11.8[(1.19x10^4\ m^3)(0.77\ m^3/sec)]^{0.56}(0.21\ tons/acre\text{-}year)(0.179)(0.5)(1.0)$$

$$= 36.7\ mt$$

The adsorbed and dissolved fractions can be calculated as follows. First, the total mass of contaminant, Mm', is divided into the adsorbed and dissolved portions using Equations (E-15) and (E-16).

Benzo(a)pyrene

$$\begin{aligned} Aa &= [1/(1+(\theta/KdB))]\ Mm' \\ &= [1/1+(0.2/(3000\ cm^3/g)(1.5\ g/cm^3))]\ 7.473\ x\ 10^{-3}\ kg/ha{\bullet}cm \\ &= 7.473\ x\ 10^{-3}\ kg/ha{\bullet}cm \end{aligned}$$

$$\begin{aligned} Da &= [1/(1+(KdB/\theta))]\ Mm' \\ &= [1/(1+((3000\ cm^3/g)(1.5\ g/cm^3)/(0.2))]\ 7.473\ x\ 10^{-3}\ kg/ha{\bullet}cm \\ &= 0.00\ kg/ha{\bullet}cm \end{aligned}$$

Cadmium

$$\begin{aligned} Aa &= [1/(1+(\theta/KdB))]\ Mm' \\ &= [1/(1+((300\ cm^3/g)(1.5\ g/cm^3))]\ 0.2332\ kg/ha\bullet cm \\ &= 0.2331\ kg/ha\bullet cm \end{aligned}$$

$$\begin{aligned} Da &= [1/(1+(KdB/\theta))]\ Mm' \\ &= [1/(1+((300\ cm^3/g)(1.5\ g/cm^3)/(0.2))]\ 0.2332\ kg/ha\bullet cm \\ &= 0.0001\ kg/ha\bullet cm \end{aligned}$$

Based on these calculations, virtually all the contaminant loss for both benzo(a)pyrene and cadmium is in the adsorbed form. The losses for each route, adsorbed (P_{xt}) and dissolved P_{qt}), are defined by Equations (E-17) and (E-18) as follows:

Benzo(a)pyrene

$$\begin{aligned} P_{xt} &= [10^{-4}X_e/(WA_x)(B)]\ Aa \\ &= [(10^{-4})(36.7\ mt)/(1\ km^2)(1.5\ g/cm^3)]\ 7.473 \times 10^{-3}\ kg/ha\bullet cm \\ &= 1.828 \times 10^{-5}\ kg/ha \end{aligned}$$

$$\begin{aligned} P_{qt} &= [D_R/(R_t + M_t)]\ Da \\ &= [1.19\ cm/(5\ cm + 0\ cm)]\ 0.00\ kg/ha\bullet cm \\ &= 0.00\ kg/ha \end{aligned}$$

Cadmium

$$\begin{aligned} P_{xt} &= [10^{-4}X_e/(WA_x)(B)]\ Aa \\ &= [(10^{-4})(36.7\ mt)/(1\ km^2)(1.5\ g/cm^3)]\ 0.2331\ kg/ha\bullet cm \\ &= 5.7032 \times 10^{-4}\ kg/ha \end{aligned}$$

$$\begin{aligned} P_{qt} &= [D_R/(R_t + M_t)]\ Da \\ &= [1.19\ cm/(5\ cm + 0\ cm)]\ 0.0001\ kg/ha\bullet cm \\ &= 2.3800 \times 10^{-5}\ kg/ha \end{aligned}$$

Once P_{xt} and P_{qt} are known, the receiving water concentrations due to adsorbed and dissolved contaminant, respectively, can be calculated from Equations (E-19) and (E-20) as follows:

Benzo(a)pyrene

$$Ci_x = \frac{(P_{xt})(WA_x)}{VS_x}$$

$$= \frac{(10^8)(1.828 \times 10^{-5}\ \text{kg/ha})(1\ \text{km}^2)}{(8.63 \times 10^7\ 1)}$$

$$= 2.1182 \times 10^{-5}\ \text{mg/l}$$

$$C_{ix} = \frac{(P_{qt})(WA_x)}{V_{sx}}$$

$$= \frac{(10^8)(0.00\ \text{kg/ha})(1\ \text{km}^2)}{(8.63 \times 10^7\ 1)}$$

$$= 0.00\ \text{mg/l}$$

Cadmium

$$Ci_x = \frac{(P_{xt})(WA_x)}{VS_x}$$

$$= \frac{(10^8)(5.7032 \times 10^{-4}\ \text{kg/ha})(1\ \text{km}^2)}{(8.63 \times 10^7\ 1)}$$

$$= 6.6086 \times 10^{-4}\ \text{mg/l}$$

$$= \frac{(P_{xt})(WA_x)}{VS_x}$$

$$= \frac{(10^8)(2.3800 \times 10^{-5} \text{ kg/ha})(1 \text{ km}^2)}{(8.63 \times 10^7 \text{ l})}$$

$$= 2.7578 \times 10^{-5} \text{ mg/l}$$

The Ci_x for cadmium represents the maximum receiving water concentration to which humans would be exposed.

E.6 REFERENCES FOR APPENDIX E

1. U. S. Environmental Protection Agency. Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge. Draft Final Report. 1986.

2. Wischmeier, W. H., and D. D. Smith. Predicting Rainfall Erosion Losses - A Guide to Conservation Planning. USDA Handbook No. 537. 1978.

3. Reference 2.

4. Reference 1.

5. Williams. F. R. Sediment and Yield Prediction with Universal Equating Using Runoff Energy Factor. Present and Prospective Technology for Predicting Sediment Yields and Sources. USDA ARS-S-40. 1975.

6. Haith, D. A. A Mathematical Model for Estimating Pesticide Losses in Runoff. J. of Environ. Qual. 9:428-433. 1980.

7. Reference 2.

8. Reference 1.

9. Bossert, D. D., and R. Bartha. Structure - Biodegradability Relationships of Polycyclic Aromatic Hydrocarbons in Soil. Bulletin of Environ. Contam. Toxicol. 37:490-495. 1986.

APPENDIX F
GROUNDWATER INFILTRATION MODEL

APPENDIX F
GROUNDWATER INFILTRATION MODEL

F.1 INTRODUCTION

Contaminants associated with particulates emitted from municipal waste combustors are subject to deposition on surfaces downwind from the facility. This fallout is subsequently subject to dissolution in rain or meltwater from precipitation events. The dissolved portion can follow one of two pathways: either move over the surface as runoff to a surface water body or infiltrate into the ground and recharge the groundwater. As a consequence, persons using the groundwater may be exposed to groundwater transported contaminants. Aquatic life inhabiting surface water bodies fed by the contaminated aquifer could be exposed as well.

The methodology derived to calculate exposure from the groundwater pathway was originally developed to evaluate impacts from the landfilling of municipal sludge and to evaluate the groundwater pathway associated with the application of sludges to land. A detailed discussion is available in the document entitled "Development of Risk Assessment Methodology for Municipal Sludge Landfilling".[1] It is formulated in three successive tiers, which begin with simple but very conservative estimates and proceed to more detailed analyses if the first tiers predict unacceptable risks. Only chronic exposure is evaluated using standard approaches to calculate leachate generation and associated groundwater transport in the unsaturated zone.

F.2 ASSUMPTIONS

A number of assumptions were required to formulate the risk-based methodology both with respect to leachate generation and subsequent transport in the unsaturated zone. The key assumptions along with a discussion of their impacts on the methodology are provided in Table F-1.

TABLE F-1. ASSUMPTIONS FOR THE GROUNDWATER PATHWAY METHODOLOGY

Functional Area	Assumption	Ramifications
Source term	Equilibrium will be reached wherein annual inputs from fallout are lost in recharge as leachate	Overpredicts to the extent that some contaminants are irreversibly bound to solids and neglects other loss mechanisms such as runoff.
	Leachate pattern is level over the entire year.	Underpredicts peak concentrations but provides estimates of average annual loading.
Unsaturated zone transport	One-dimensional flow in the vertical direction.	Overpredicts concentration since it ignores horizontal dispersion.
	Water flow is steady-state.	Overpredicts concentration by accelerating flow in a compressed time period.
	Upper boundary has a constant flux of recharge.	Overpredicts or underpredicts, depending on the soil type.
	Soil characteristics are constant with depth for any layer.	It is impossible to determine the effect of this assumption, since it will vary from site to site.
	Vertical hydraulic gradient of unity (not assumed in one of two alternative approaches).	Overpredicts concentration to the extent that gradients may be <1.0 and, therefore, time of travel is slower, allowing for more degradation.

TABLE F-1. (Continued)

Functional Area	Assumption	Ramifications
Unsaturated zone transport (cont.)	Retardation of organics is related to soil organic fraction only.	Overpredicts contaminant velocity for soils with low organic content where mineral interaction may predominate.
	All adsorption is reversible.	Overpredicts concentration arriving at aquifer.
	First-order degradation mechanism.	Mispredicts degradation where higher order rates are functional: Overpredicts zero order mechanisms.
Saturated zone (MINTEQ Calculation)	Groundwater conditions dictate geochemistry.	Effects would be site-specific depending on the quantity and quality of both the leachate and the aquifer.
	The six groundwater pH-Eh couplets modeled provide an adequate set of alternatives.	Some sites could have extreme conditions beyond those modeled. If pH values are very low, the model will underpredict, and if Eh conditions are very low, the model will overpredict contaminant concentrations.
	Contaminant interactions do not affect geochemistry.	Effects would be source-specific.

Due to the inexact nature of the science, any assumptions made are conservative (i.e., overpredict contaminant concentrations). In some instances, however, the nature of the effect of a given assumption may vary with site-specific considerations.

F.3 CALCULATIONS

The methodology for evaluating the groundwater pathway is illustrated in Figure F-1. Tier 1 involves projecting leachate concentrations. Leachate concentrations are predicted on the basis of annual fallout and recharge rates. It is assumed that soil contaminant levels will increase over time until leachate strength is sufficiently high to deplete the input from fallout each year. Hence, leachate concentration is defined as:

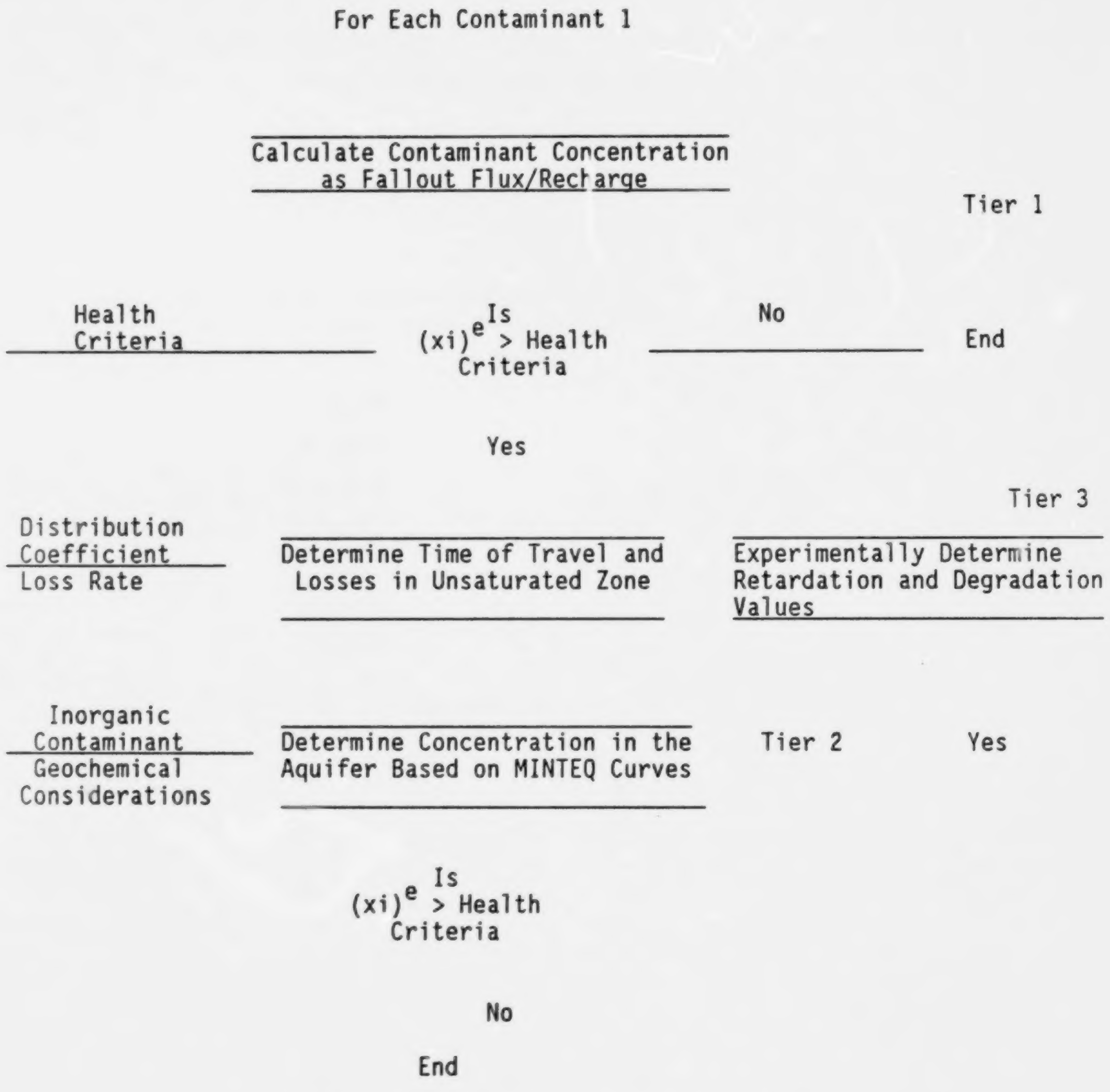
$$X_i = Fa/R_c \qquad \text{Equation (F-1)}$$

where:

X_i = leachate contaminant concentrations (M/L^3)
Fa = annual fallout flux of contaminant/unit area (M/L^2-T)
R_c = annual depth of recharge (L/T).

Any contaminant found to have an insignificant leachate concentration can be eliminated from further consideration. Those with high leachate concentrations are carried forward to a Tier 2 or 3 analysis. Currently, the methodology can simulate geochemistry for seven metal contaminants (Table F-2). For other inorganics, it is necessary to assume that no precipitation reactions will occur.

The annual mass of contaminant fallout (Fa) should be calculated as the average value that falls within a 200 m radius of the combustor. The 200 m distance was chosen because it is the closest to the combustor that model estimates can make accurately. In addition, the values at 200 m showed the greatest deposition (most conservative). Also, a watershed of this size is large enough to provide a potable water supply to several families (based on a recharge rate of 0.25 m/year and the assumption that each person uses 100 gallons/day). Therefore, it is not unreasonable to assume that a single well can be supplied by an area this size.


For Each Contaminant 1
Calculate Contaminant Concentration as Fallout Flux/Recharge
Tier 1
Health Criteria
Is $(xi)^e$ > Health Criteria
No
End
Yes
Tier 3
Distribution Coefficient Loss Rate
Determine Time of Travel and Losses in Unsaturated Zone
Experimentally Determine Retardation and Degradation Values
Inorganic Contaminant Geochemical Considerations
Determine Concentration in the Aquifer Based on MINTEQ Curves
Tier 2
Yes
Is $(xi)^e$ > Health Criteria
No
End

$(xi)^e$ = Concentration of Contaminant 1

FIGURE F-1. LOGIC FLOW FOR GROUNDWATER PATHWAY EVALUATION

TABLE F-2. METAL CONTAMINANTS SIMULATED IN THE GEOCHEMICAL PORTION OF THE GROUNDWATER PATHWAY

1.	Arsenic
2.	Cadmium
3.	Chromium
4.	Copper
5.	Lead
6.	Mercury
7.	Nickel

Presumably, the compliance point falls within the zone of deposition. Hence, no saturated zone calculations are performed. The compliance point is assumed to be that point at the base of the unsaturated zone where leachate enters the saturated zone. For metals, the contaminant concentration in the groundwater is adjusted based on geochemical reactions. For organics, the contaminant concentration is adjusted due to degradation as it travels through the unsaturated zone.

The Tier 2/3 analysis begins at the same point as Tier 1, calculation of the source-term strength of the leachate from the fallout. Tier 2/3, however, allows for site-specific inputs to predict dispersion, degradation, and retardation effects, which reduce resultant exposure levels. The difference between Tiers 2 and 3 lies in the number of input parameters, which are determined experimentally. In Tier 3, degradation rates and retardation coefficients are measured directly.

The initial step in Tier 2/3 is to define the strength of contaminant in recharge water. As with Tier 1, this is done by assuming that equilibrium will be reached wherein the inputs from fallout each year will be transported away from the soil in leachate. Equation (F-1) is used to calculate the average leachate contaminant concentration over the life of the facility and its replacements.

For degradable contaminants, Equation (F-1) is modified to :

$$X_i = Fa(1-e^{-kt})/t\ k\ R_c \qquad \text{Equation (F-2)}$$

where:

X_i = average leachate contaminant concentration (M/L^3)
Fa = fallout contaminant flux (M/L^2-T)
k = first-order degradation rate for contaminant (T^{-1})
t = 1 year (T)
R_c = annual depth of recharge (L/T)

The next step is to calculate the time required for leachate thus formed to move downward through the unsaturated zone to the aquifer. This is accomplished by making a time of travel calculation with one of two analytic approaches.[2] The selected approach assumes constant moisture content in the soil with steady-state flow and a unit hydraulic gradient. It uses the soil moisture, pressure and conductivity relationships described by Campbell to solve for the moisture content of the soil:[3]

$$f = fs(q/K_{SAT})^{1/(2b+3)} \qquad \text{Equation (F-3)}$$

where:

f = field moisture content (L^3/L^3)
q = moisture flux, recharge rate in this case (L/T)
K_{SAT} = saturated hydraulic conductivity of the soil in the unsaturated zone (L/T)
fs = saturated moisture content of the soil (L^3/L^3)
b = negative one times the slope of the log-log plot of matric potential and saturated moisture content (dimensionless)

The velocity and travel times for flow in the unsaturated zone are related according to:

$$V = q/f \qquad \text{Equation (F-4)}$$

$$T = hy/V = (hy)(f)/q \qquad \text{Equation (F-5)}$$

where:

V = velocity of flow (L/T)
T = time of travel (T)
hy = depth of the unsaturated zone (L)

For multiple layer systems, a travel time is calculated for each layer and the total summed across the unsaturated zone. The total time is then divided into the total depth to derive the equivalent velocity.

The velocity calculated from Equation (F-4) is for water traveling through the unsaturated zone. If the contaminant interacts with the soil in transit, it will travel at a retarded velocity, V_R, defined as:

$$V_R = V/[1 + ((B/\theta)(Kd))] \qquad \text{Equation (F-6)}$$

where:

V_R = velocity of the contaminant (L/T)
V = velocity of water calculated from Equation (F-4)

B = average bulk density of soil in the unsaturated column (M/L^3)

θ = average porosity of soil in the unsaturated column (dimensionless)

Kd = soil-water partition coefficient for contaminant (L^3/M)

If the contaminant is degradable, its concentration will change according to:

$$X_a = X_i \, e^{[-k(hy)/V_R]} \qquad \text{Equation (F-7)}$$

where:

X_a = contaminant concentration upon entry in the aquifer (M/L^3)

X_i = concentration of leachate initially from Equation (F-2)

k = first-order degradation rate for contaminant (T^{-2})

hy = depth of unsaturated zone (L)

V_R = retarded velocity form Equation F-6 (L/T)

e = base of natural logarithms, 2718 (unitless)

If the analyst wishes to account for dispersion as well as attenuation and degradation in the unsaturated zone, the water velocity V from Equation (F-4) can be input to the one-dimensional CHAIN code.[4] The CHAIN code is an analytical solution of the convective-dispersive transport equation for a one-dimensional case that accounts for retardation and degradation. The data inputs to the model include the average pore water velocity, the dispersion coefficient, the water content, the pulse or release time, the retardation factor, the decay rate, and several coefficients describing the source term. The output from the code is the concentration of the contaminant plume at the base of the unsaturated zone for a period of time equal to several contaminant travel times through the unsaturated zone.

For organic contaminants, the concentration calculated from Equation (F-7) is the predicted concentration at the compliance point. For metals, geochemistry can be considered using a series of input-output curves generated by the MINTEQ code, which establishes the solubility limits for a contaminant metal on the basis of groundwater (ionic) composition, pH, and Eh conditions. The MINTEQ geochemical code can be applied to generate predicted contaminant concentrations under selected groundwater conditions.

MINTEQ is a hybrid code that combines an efficient mathematical structure with a large, well documented thermodynamic data base. Functionally, the code models the mass distribution of a dissolved element between various uncomplexed and complexed aqueous species; it also calculates the degree to which the water is saturated with respect to the solids in the thermodynamic data base. Adsorption, precipitation, and dissolution reactions can be included in calculations. Detailed documentation of the MINTEQ code and data can be found in Felmy et al. and Deutsch and Krupka.[5,6,7,8] Curves for selected pH-Eh couplets and an explanation of how the relations were derived can be found in U.S. EPA.[9] An example MINTEQ curve is shown in Figure F-2.

Output from the unsaturated zone [Equations (F-6) and (F-7) or the CHAIN code and the MINTEQ input-output curves] are the predicted contaminant concentration and timing at the compliance point. The pulse time is assumed continuous for the period of operation of the combustor. If the output from the Tier 2 analysis exceeds exposure-based criteria at the compliance point, the analyst may choose to initiate a Tier 3 evaluation of inputs.

F.4 REQUIRED INPUTS

For a Tier 1 analysis, the only data required are the contaminant deposition rate and the recharge rate.

Both Tier 2 and Tier 3 have the same input parameter requirements. The basic difference is that Tier 3 uses more site-specific data than Tier 2. In Tier 2, data on hydraulic conductivity, recharge, depth to groundwater and soil type should be determined empirically from samples. Other soil-related properties can be selected on the basis of soil type, while degradation rate constants and partition coefficients can be taken from the literature. For Tier 3, the latter properties should also be determined empirically with samples from the site.

The input parameters required for a Tier 2/3 analysis are shown in Table F-3.

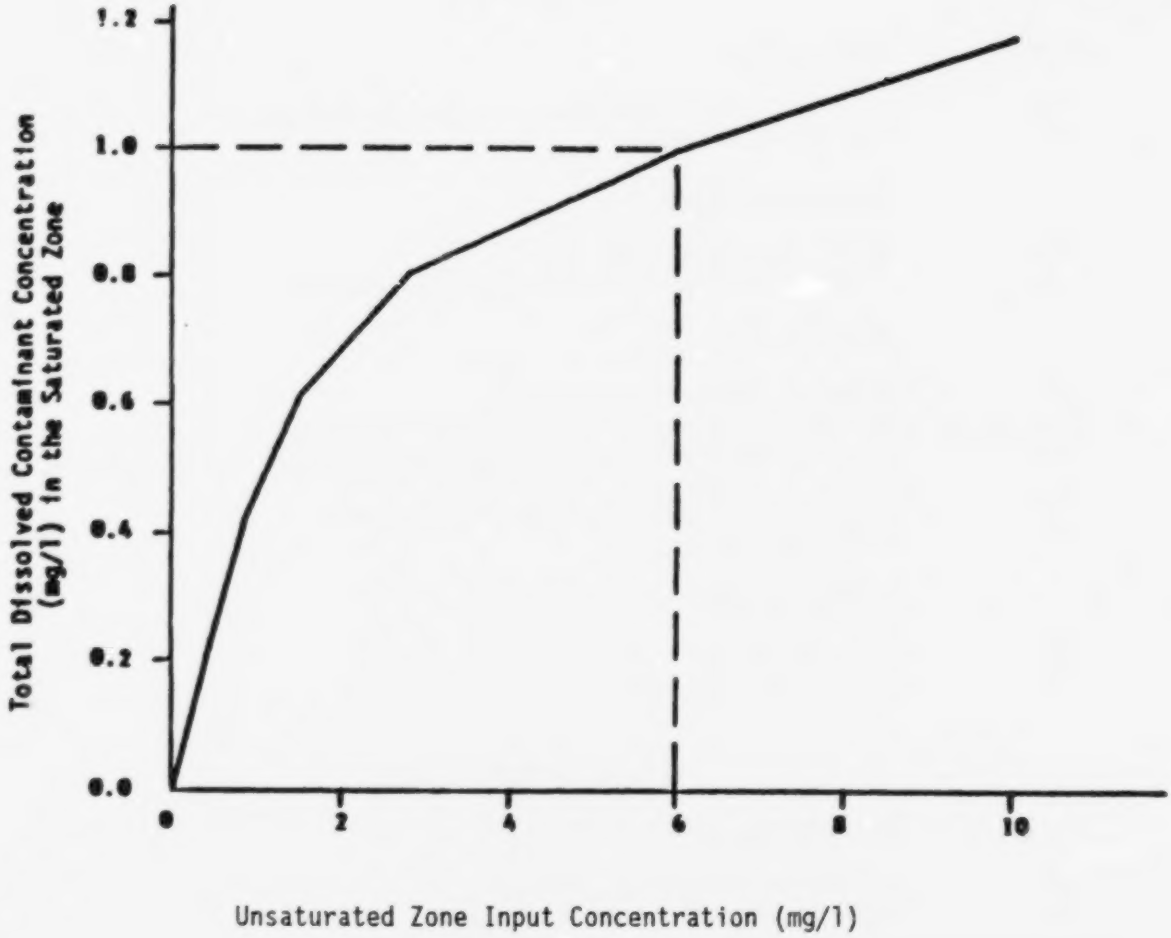


FIGURE F-2. EXAMPLE MINTEQ SPECIATION RESULTS FOR ENTRY OF A CONTAMINANT INTO THE SATURATED ZONE FOR CONDITIONS OF pH = 7.0 AND Eh = 1.50 mv

TABLE F-3. INPUT PARAMETERS FOR GROUNDWATER PATHWAY

	PATHWAY DATA
Symbol	Source Data
Fa	flux rate for contaminant fallout (mg/m^2-year)
Rc	net recharge (m/year)
	Unsaturated Zone
hy	depth to groundwater (m)
X_1	leachate contaminant concentration (mg/m^3)
	Material Type
m	Material layer thickness (m)
K_{SAT}	saturated hydraulic conductivity (m/year)
b	slope of matrix potential and moisture content plot (dimensionless)
fs	saturated soil moisture content (m^3/m^3)
B	bulk density (kg/m^3)
	Saturated Zone
pH	groundwater pH
Eh	groundwater Eh (mvolts)
	CHEMICAL-SPECIFIC DATA
	Unsaturated Zone
Kd	partition coefficient (ℓ/kg)
k	degradation rate constant ($year^{-1}$)

F.5 EXAMPLE CALCULATIONS

The methodology presented above can best be illustrated with example calculations. A site-specific example calculation is provided below for two contaminants, cadmium and benzo(a)pyrene. For both cases, a Tier 1 and a Tier 2/3 analysis are illustrated.

All input parameters for the example calculation are shown in Table F-4. Data values are for illustrative purposes only but are, for the most part, representative of a real site. The annual mass of contaminant fallout (Fa) was calculated by averaging electrostatic precipitator measurements over a 5-year period. Measurements were conducted along concentric rings at ranges varying from 200-50,000 m from the combustor on rays spaced every 22.5°. The average Fa at the 200 m radium was chosen for the reasons discussed in Section F-3.

F.5.1 Tier 1

For a Tier 1 analysis, the leachate contaminant concentration is calculated using Equation (F-1) as follows:

Benzo(a)pyrene

$$\begin{aligned} X_i &= Fa/R_c \\ &= (0.33494 \text{ mg/m}^2\text{-year})/(0.25 \text{ m/year}) \\ &= 1.340 \text{ mg/m}^3 \end{aligned}$$

Cadmium

$$\begin{aligned} X_i &= Fa/R_c \\ &= (6.42288 \text{ mg/m}^2\text{-year})/(0.25 \text{ m/year}) \\ &= 15.692 \text{ mg/m}^3 \end{aligned}$$

TABLE F-4. INPUT PARAMETERS FOR THE EXAMPLE CALCULATIONS

Rx	Annual depth of recharge, 0.25 m/year
Fa	Fallout contaminant flux: benzo(a)pyrene - 0.56642 mg/m^2-year cadmium - 10.880992 mg/m^3-year
fs	Saturated moisture content of soil, 0.4 m^3/m^3
q	Moisture flux recharge rate, 0.25 m/year
K_{SAT}	Saturated soil hydraulic conductivity, 10^4 m/year
b	Slope of matrix potential and moisture content plot, 4.0
hy	Depth of unsaturated zone, 2 m
B	Soil bulk density, 1.5 g/cm^3
θ	Soil porosity, 0.2
Kd	Soil-water partition coefficient: benzo(a)pyrene - 3000 cm^3/g cadmium - 300 cm^3/g
k	First-order degradation rate: benzo(a)pyrene - 0.16 year^{-1} cadmium - 0.00 year^{-1}
pH	Groundwater pH, 8.0
Eh	Groundwater Eh, -200 mv

F.5.2 Tier 2/3

A Tier 2/3 analysis begins at the same point as Tier 1, calculation of the source-term strength of the leachate from the fallout. Equation (F-1) is used to calculate the average leachate concentration over the life of the facility.

For cadmium, which is nondegradable, Equation (F-1) is used to calculate the average leachate contaminant concentration just as in Tier 1 above. For benzo(a)pyrene, which is degradable, Equation (F-2) is used to calculate the average leachate contaminant concentration as follows:

$$X_i = \frac{Fa(1-e^{-kt})}{(t)(k)(R_c)}$$

$$= \frac{(0.33494\ mg/m^2\text{-}year)[1-e^{-(0.16/year)(1\ year)}]}{(1\ year)(0.16/year)(0.25\ m/year)}$$

$$= 1.238\ mg/m^3$$

The next step is to calculate the time required for leachate to move downward through the unsaturated zone. First, the constant moisture content is calculated using Equation (F-3):

$$f = fs\ \frac{q}{K_{SAT}}^{1/(2b+3)}$$

$$= \frac{0.4\ m^3/m^3\ 0.25\ m/year}{10^4\ m/year}^{1/[2(4)+3]}$$

$$= 0.15$$

The water velocity travel time through the unsaturated zone can be calculated as follows:

$$T = (hy)(f)/q$$
$$= (2\ m)\ (0.15)/(0.25\ m/year)$$

$$= 1.2\ year$$

Since both benzo(a)pyrene and cadmium travel with a retarded velocity compared to the groundwater, their velocities through the unsaturated zone can be calculated with Equation (F-6) as follows:

Benzo(a)pyrene

$$
\begin{aligned}
V_R &= V/[1+((B/\theta)(Kd))] \\
&= 1.67 \text{ m/year}/[1+((1.5 \text{ g/cm}^3/0.2)(3000 \text{ cm}^3/\text{g}))] \\
&= 7.42 \times 10^{-5} \text{ m/year}
\end{aligned}
$$

Cadmium

$$
\begin{aligned}
V_R &= V/[1+((B/\theta)(Kd))] \\
&= 1.67 \text{ m/year}/[1+((1.5 \text{ g/cm}^3/0.2)(3000 \text{ cm}^3/\text{g}))] \\
&= 7.42 \times 10^{-4} \text{ m/year}
\end{aligned}
$$

Since benzo(a)pyrene is degradable, its concentration will change as it travels through the unsaturated zone. The benzo(a)pyrene concentration at the base of the unsaturated zone before it enters the aquifer can be calculated with Equation (F-7) as follows:

$$
\begin{aligned}
X_a &= X_i \, e^{[-k(hy)/V_R]} \\
&= 1.238 \text{ mg/m}^3 \, e^{[-(0.16/\text{year})(2 \text{ m})/(7.42 \times 10\text{-}5 \text{ m/year})]} \\
&= 0.00 \text{ mg/m}^3
\end{aligned}
$$

Because the benzo(a)pyrene travel time through the unsaturated zone is so long, it degrades before it reaches the aquifer. The cadmium will not degrade, but travels much slower (1800 times slower) than water. Assuming no dispersion, the cadmium concentration may still be reduced due to precipitation reactions in the saturated system. The MINTEQ results for cadmium for pH and Eh values close to the values specified in the example problem are shown in Figure F-3. An unsaturated zone concentration of 0.025 mg/l for cadmium (Y axis in Figure F-3) corresponds to a concentration of 0.019 mg/l (19 ug/l) in the saturated zone after accounting for speciation.

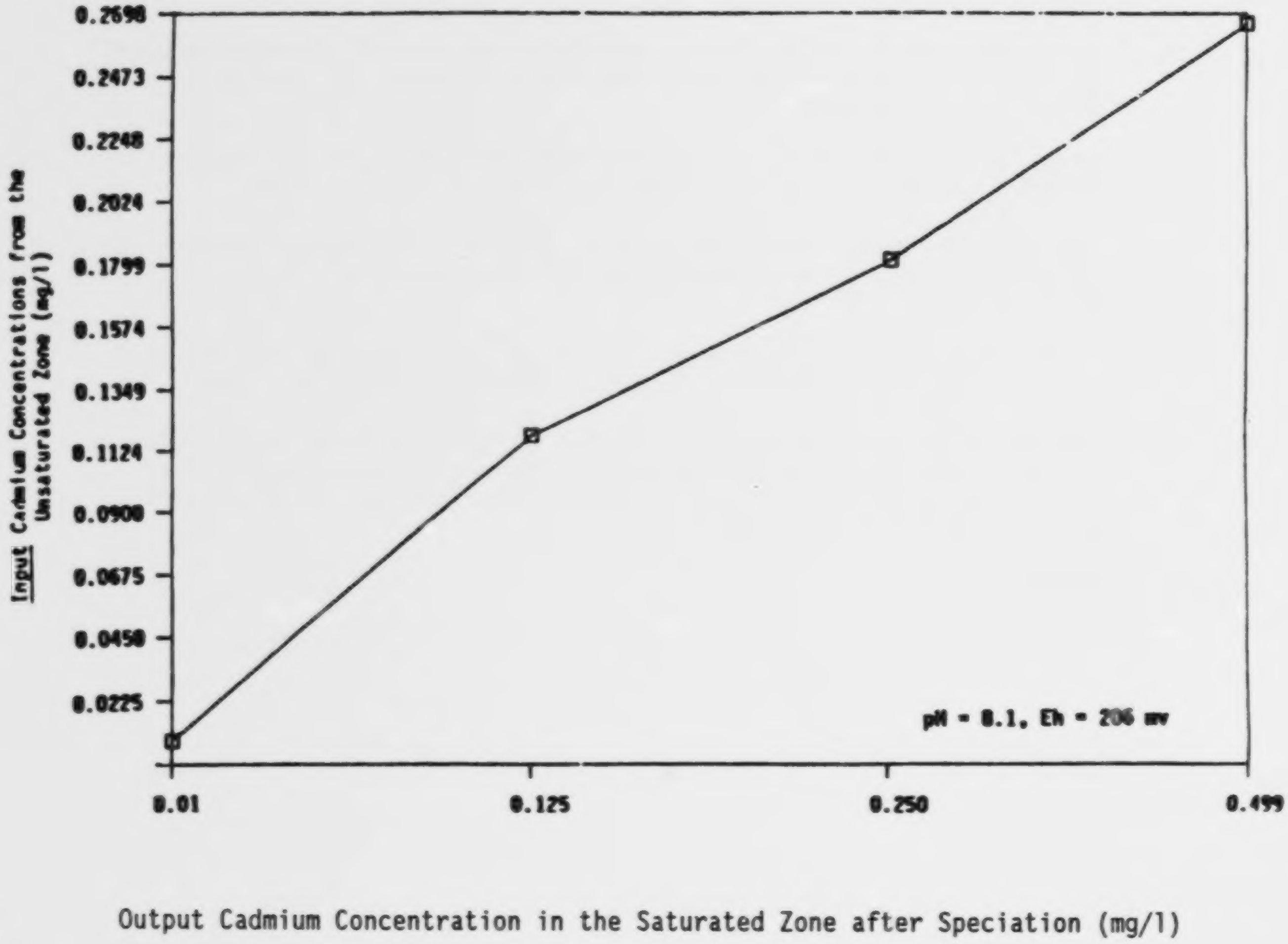


FIGURE F-3. GROUNDWATER CADMIUM SPECIATION

F.6 REFERENCES FOR APPENDIX F

1. U.S. Environmental Protection Agency. Development of Risk Assessment Methodology for Municipal Sludge Landfilling. Draft Final Report. 1986.

2. U.S. Environmental Protection Agency. Technical Support for Development of Guidance on Hydrogeologic Criterion for Hazardous Waste Management Facility Location. Draft Report. 1985.

3. Campbell, G.S. A Simple Method for Determining Unsaturated Conductivity from Moisture Retention Data. Soil Science 117: 311-314. 1974.

4. Van Genuchton, M. Concentive-dispersive transport of Solutes Involved in Sequential First-order Decay Reactions. Journal of Computers Geosci. 11: 129-147. 1985.

5. Felmy, A.R., S.M. Brown, Y. Omishi, S.B. Yabusaki, and R.S. Argo. MEXAMS - The Metals Exposure Analysis Modeling System. 1983.

6. U.S. Environmental Protection Agency. MINTEQ - A Computer Program for Calculating Aqueous Geochemical Equilibria. EPA Report No. 600/3-84-032. 1984

7. Morrey, J.R. PRODEF - A Code to facilitate the Use of the Geochemical Code MINTEQ. Draft Report. Battelle Memorial Institute. 1985.

8. Deutsch, W.J. and K.M. Krupka. MINTEQ Geochemical Code: Compilation of Thermodynamic Database for the Aqueous Species, Gases, and Solids Containing Chromium, Mercury, Selenium, and Thallium. Draft Report. Battelle Memorial Institute. 1985.

9. Reference 1.

APPENDIX G
DERMAL EXPOSURE MODEL

APPENDIX G
DERMAL EXPOSURE MODEL

G.1 INTRODUCTION

The dermal exposure model refers to human skin contact with contaminants from emissions of municipal waste combustors deposited on the soil. The issue of dermal absorption of deposited contaminants is very complex. There is a fundamental lack of data for percutaneous absorption of chemicals in human skin from soil. Other factors important for estimation of human exposure to contaminants by the dermal route also have many uncertainties. The model described here is offered as a possible approach for the estimation of human exposure and risk associated with dermal exposure, but it recognized that in most, if not all cases, the available data will not provide a satisfactory basis for risk calculations.

G.2 MOST-EXPOSED INDIVIDUALS (MEIs)

The MEI for this pathway of dermal exposure is an individual residing within 50 km of a municipal waste combustor (in the area of maximal deposition of emissions) who spends a majority of this daily activity outdoors. Preschool children (between 1-6 years of age) who play outdoors or rural farmers would most likely be the MEIs, since these groups have the greatest opportunity for dermal exposure to particulates deposited on the soil. These children are likely to be exposed in residential areas (gardens, lawns, parks, etc.). Farmers or individuals who garden would also have the potential for substantial skin contact with soil. Occupational exposures of workers involved in the operation of municipal waste combustors are not considered here since these workers can be required to use special measures or equipment to minimize their exposure to possibly hazardous materials.

G.3 DEPOSITION-HUMAN ("DERMAL") TOXICITY EXPOSURE PATHWAY

G.3.1 Assumptions

For this methodology, it will be assumed that the daily dermal intake of the contaminant increases continually with contaminant concentration in soil particles contacting the skin. It is assumed that 100 percent of the deposited contaminant particulate is retained in the uppermost 1 cm layer of soil. The maximum concentration of deposited contaminant within 50 km of a municipal waste combustor is used. It is further assumed that this soil layer contacts the exposed skin of individuals involved in outdoor activities. Consideration of dermal exposure during outdoor activity to contaminated particulate deposited on soil most likely accounts for a more substantial exposure to the individual than contaminated particulate deposited in indoor (residential) dust.[1] Indoor dust is therefore excluded from this model at the present time.

Dermal intake of contaminants is assumed to be a function of the fraction of the compound absorbed, contact time (or duration of daily exposure), exposed skin surface area, contact amount (amount of soil accumulated on skin) and soil contaminant concentration.

Calculation of the daily intake by dermal exposure is the same for organics and inorganics, since there is immediate exposure potential and therefore no soil incorporation. Background concentration of contaminants is not included because this methodology assesses only the risk associated with the increase in exposure due to the contaminant from municipal waste combustor emissions. The assumptions and uncertainties relevant to this model and their ramifications/limitations are shown in Table G-1.

G.3.2 Calculation Method

A daily dermal intake (DDI, ug/day) is determined as follows:

$$DDI = Ct \times SA \times CA \times AF \times LC_T \times (10^{-3}/24) \times EDA \qquad \text{Equation (G-1)}$$

TABLE G-1. ASSUMPTIONS AND UNCERTAINTIES FOR DERMAL EXPOSURE MODEL

Functional Area	Assumption/Uncertainty	Ramifications/Limitations
Exposure assessment	Deposited emissions are not necessarily soil incorporated and may concentrate in the uppermost soil layer. Deposited contaminant is assumed to be distributed within the uppermost 1 cm of soil, and the soil which contacts the skin is assumed to originate from the same 1 cm layer.	Overestimates exposure in situations where soil incorporation occurs to any depth >1 cm.
Dermal intake	Linear with respect to soil concentration.	May overestimate exposure.
Fraction of contaminant absorbed	Chemical and matrix specific.	Uncertainty for many contaminants and matrices.
	Data needed for dermal absorption in humans may not be available.	Uncertainty in between species extrapolation for dermal exposure. Uncertainty in route-to-route extrapolation of absorption.
	Dermal absorption varies with age.	Absorption data for a contaminant at one age may over- or underestimate actual absorption at another concentration.
	May be concentration dependent.	Study data at one concentration may over- or underestimate actual absorption at another concentration.
Contact time	The MEIs are exposed to contaminated particulate 12 hours/day during outdoor activity.	May underestimate exposure by excluding indoor exposure to dust.

TABLE G-1. (Continued)

Functional Area	Assumption/Uncertainty	Ramifications/Limitations
Exposed skin surface area	Total intake is proportional to exposed surface area; adults = 2940 cm^2; children = 980 cm^2.	May over- or underestimate intake.
Soil contact amount	Children = 1.5 mg/cm^2; Adults = 1.5 - 3.5 mg/cm^2.	May over- or underestimate exposure.
Effects assessment	Substantial dermal exposure may occur only from 1 to 6 years of age. Cancer potency is adjusted to reflect a 5-year rather than 70-year exposure.	Hazard could be underestimated if child is more susceptible to chemical carcinogenesis than an adult and 5/70 is used as an adjustment factor.
	Lifetime exposure may need to be adjusted for days/year in contact with contaminant during 70-year lifetime.	Uncertainty as to appropriate adjustment.

where:

DDI = human daily dermal intake (ug/day)
CT = contact time (hours/day)
SA = exposed skin surface area (cm^2)
CA = contact amount (mg/cm^2)
AF = absorption fraction (%/day)
LC_T = maximal soil concentration at time, T (ug/g)
EDA = exposure duration adjustment (unitless)
$(10^{-3}/24)$ = conversion factor (10^{-6} x 1 day x 10^3 ug)(ug x 24 hours x mg)

The DDI represents the increase (above background) in daily human dermal intake (ug/day) due to the contaminant from municipal waste combustor emissions.

G.3.3 Input Parameter Requirements

G.3.3.1 Absorption fraction (AF). Dermal absorption of a contaminant is both chemical-specific and matrix-dependent.[2,3] Physicochemical properties of the contaminant (e.g., lipid solubility) will affect dermal absorption. Factors such as pH, molecular size, temperature and humidity will also influence absorption. Absorption of a contaminant in a matrix or vehicle is affected by the physicochemical properties of the matrix. There is uncertainty as to how various soil or particulate types or matrices would influence absorption of deposited contaminants. Poiger and Schlatter demonstrated that a soil matrix reduced the absorption of TCDD when TCDD was applied to the skin of rats in a soil-water paste.[4]

Dermal absorption in this case refers to the fraction of the applied dose of the compound absorbed by human skin within a day (i.e., 6%/ 24 hours). Such data for a contaminant in a particulate or soil matrix should be used but are rarely available. Absorption studies of the contaminant by the dermal route of exposure may not be available, while exposure by other routes may be present in the literature. Guidelines, however, for route-to- route extrapolation of absorption of toxicants have not been clearly delineated and pose uncertainties, especially where dermal absorption is concerned. Extrapolation of percutaneous absorption data among animal species would introduce additional uncertainty.

Absorption of contaminants may be altered if the skin is damaged (diseased, lacerated, or abraded).[5] Percutaneous absorption may also vary with age. These issues pose uncertainties and need further research.

The fraction of the contaminant absorbed may vary with concentration.[6] The assumption of complete absorption of a contaminant irrespective of dose is the most conservative approach (AF = 1), but may be unrealistic. For example, Kimbrough et al. suggested human dermal absorption of TCDD from soil was ~1 percent.[7]

The estimation of the fraction absorbed of a contaminant is very complex and dependent on many factors as discussed above. A paucity of data regarding the dermal absorption of chemicals in humans, particularly in particulate or soil matrices, will make estimation of the fraction of contaminant absorbed from soil difficult for this methodology.

G.3.3.2 Contact Time (CT). Contact time is defined as the amount of time/day the MEIs would spend in association with contaminated soil. Children playing outdoors and adult farmers would contact soil very frequently, and soil contact would continue until the cleansing of exposed skin. The maximum contact time for these individuals will be assumed to be 12 hours/day.[8] Contact time would probably be greater if indoor dust exposure was also considered.

G.3.3.3 Surface Area (SA). Total intake of a contaminant would be approximately proportional to the exposed surface area for absorption. The anatomical region and surface area of skin that is directly exposed to the contaminant will affect dermal intake of the contaminant since there is anatomical variability in percutaneous absorption.[9] Exposed surface area of adults wearing short-sleeved, open-necked shirts, pants and shoes, with no gloves or hat, is ~2940 cm^2, whereas that of children wearing the same clothing is 980 cm^2.[10]

G.3.3.4 Contact Amount (CA). The contact amount or amount of soil accumulating on skin is considered to have an upper limit of 1.5 mg/cm^2 for children[11,12] based on the reports of Lepow et al. and Roels et al.[13,14] The contact amount for children was assumed to also apply to adults. Hawley, however, suggested the soil coating on adults could be as great as 3.5 mg/cm^2.[15]

G.3.4 Example Calculations

G.3.4.1 Cadmium. The DDI for cadmium for adults can be estimated using Equation 3-38 (see Section X.2.2). The values for SA, CA and CT are 2940 cm^2, 1.5 mg/cm^2 and 12 hours/day, respectively. For the purpose of this example only, the AF will be arbitrarily set at 1%/day. The LC is 21.76 and 72.53 ug/g for 30 and 100 years, respectively.

For 30 years, DDI = 12 hours/day x 2940 cm^2 x 1.5 mg/cm^2 x 0.01/day x 21.76 ug/g x (10^{-3}/24) x 1 = 0.48 ug/day.

For 100 years, DDI = 12 hours/day x 2940 cm^2 x 1.5 mg/cm^2 x 0.01/day x 72.53 ug/g x (10^{-3}/24) x 1 = 1.60 ug/day.

G.3.4.2 Benzo(a)Pyrene. Values of DDI for B(a)P for adults and children are calculated in the same manner as above for cadmium; however, the LC for B(a)P, 1.17 x 10^{-2} ug/g for 30 and 100 years, is substituted. For this example, AF will be arbitrarily set at 1%/day. The RE is derived as the reciprocal of the absorption fraction (50%) for ingested B(a)P.[16] The EDA remains at 1 for lifetime exposure.

DDI = 12 hours/day x 2940 cm^2 x 1.5 mg/cm^2 x 0.01/day x 2.36 x 10^{-1} ug/g x (10^{-3}/24) x 1 = 5.20 x 10^{-3} ug/day

G.4 REFERENCES FOR APPENDIX G

1. Hawley, J. K. Assessment of Health Risk from Exposure to Contaminated Soil. Risk Analysis 5:289-302. 1985.

2. U. S. Environmental Protection Agency. Dermatotoxicity. EPA Report No. 560/11-82-002. 1982.

3. Reference 1.

4. Poiger, H., and C. Schlatter. Influence of Solvents and Adsorbents on Dermal and Intestinal Adsorption of TCDD. Food Cosmet. Toxicol. 18:477-481. 1980.

5. Reference 2.

6. Feldman, R. J., and H. I. Maibach. Percutaneous Penetration of Some Pesticides and Herbicides in Man. Toxicol. Appl. Pharmacol. 28:126-132. 1974.

7. Kimbrough, R. D., H. Falk, P. Stehr, and G. Fries. Health Implications of 2,3,7,8-tetrachlorodibenzodioxin (TCDD) Contamination of Residential Soil. J. of Toxicol. Environ. Health 14:47-93. 1984.

8. Reference 1.

9. Maibach, H. I., R. J. Feldman, T. H. Milby, and W. F. Serat. Regional Variation in Percutaneous Penetration in Man. Arch. Environ. Health 23:208-211. 1971.

10. U. S. Environmental Protection Agency. Risk Analysis of TCDD Contaminated Soil. EPA Report No. 600/8-84-031. 1984.

11. Reference 10.

12. Reference 1.

13. Lepow, M. L., M. Gillette, S. Markowitz, R. Robino, and J. Kashin. Investigations into Sources of Lead in the Environment of Urban Children. Environ. Res. 10:415-426. 1975.

14. Roels, H. A., J. P. Buchet, and R. R. Lauwreys. Exposure to Lead by the Oral and Pulmonary Routes of Children Living in the Vicinity of a Primary Lead Smelter. Envir. Res. 22:81-94. 1980.

15. Reference 1.

16. U. S. Environmental Protection Agency. Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: Cadmium. 1985.